Studies of Headwater Brook Charr (*Salvelinus fontinalis*) Populations in Southern Ontario affected by Groundwater Extraction

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Habitat conditions, use and preference, size-at-age, density and condition were assessed for headwater brook charr (*Salvelinus fontinalis*) populations in streams in Southern Ontario where water extraction was occurring. Research was divided into two separate, but related studies, conducted during the summer and fall of 2002 and 2004. The first study sought to document the habitat conditions that influence size-at-age and density of age-0 brook charr and to test the hypotheses that: [1] differences in fork-lengths among sites were principally influenced by differences in stream thermal regimes as measured by accumulated thermal units (ATUs), with differences in stream physical characteristics (e.g., water quality, morphology, riparian vegetation) also having significant effects; [2] among-stream differences in flow regimes would significantly influence among-stream differences in age-0 densities, with increases in flow being positively correlated with density; and, [3] length and density would be negatively correlated. Results indicated that differences in temperature, stream site and flow influenced differences in age-0 length and density of brook charr. The effect of temperature, as measured by accumulated thermal units, dominated other influences, providing strong support for the null hypothesis [1] that size differences among sites were primarily driven by differences in the prevailing thermal regime. Some evidence was found to suggest that differences in stream physical characteristics influenced fish size and density, with stream flow affecting both to a moderate degree. Thus while study results provide support for the hypothesis [2] concerning flow-related impacts on age-0 brook charr in headwater streams, changes in flow cannot be considered of prime importance over the range of stream discharge differences (14.3-28.5 $L \cdot s^{-1}$) observed here. Finally, there was no support for hypothesis [3] concerning the negative correlation between age-0 fork-length and density.

The second study investigated the effects of municipal groundwater pumping on stream habitat and the resident brook charr population of Monora Creek, Orangeville, Ontario. Specifically, the study aimed to test the hypotheses that: [1] well-related water extraction was associated with habitat degradation at sites proximate to the extraction point; and, [2] degraded habitat was associated with reduced juvenile and adult brook charr use and preference, reduced occupancy density and reduced condition. Study results indicated significant, although subtle, changes in available habitat as a result of pumping activity, with habitats in areas impacted by groundwater pumping showing significant decreases in hydraulic gradient, stream depth and the

availability of under-cut bank and run and pool habitat. Combined, the changes in habitat influenced use, with age-0 densities declining in stream reaches impacted by pumping relative to reaches of the stream not impacted by pumping. When adjusting for the relative availability of groundwater extraction on affected and non-affected habitat, standardized habitat preference indices indicated a significant preference for non-affected stream reaches among age-0 brook charr. Adults evidenced only slight changes in distribution among the available habitats that were seasonally dependent, with reduced densities observed in affected habitats and increased densities in non-affected habitats noted only in the fall. Changes in brook charr condition (weight-length relationship) and mortality were generally not in evidence during the course of this study.

Combined, these studies suggest development actions moderating either the thermal or flow regime of streams will have negative effects on population status. Water extraction risks altering habitat in unfavourable ways by interrupting important linkages between flow and temperature that favour growth and density, but that prevent critical physiological thresholds for temperature from being surpassed. Furthermore, the impacts on brook charr populations subjected to moderating pumping appear to relate more to patterns of within stream distribution and habitat use than to any direct measures of fitness such as survival or growth. The latter result suggests monitoring abundance and/or condition alone will not suffice to provide early warning of population stress responses to water extraction activities.

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Dedication

This thesis is dedicated to my father and mother, who taught me that persistence and determination can help you overcome what might seem impossible. It is also dedicated to my wife and kids, who have supported me in this long journey.

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CHAPTER 1. INTRODUCTION

Headwater streams are important habitats for fish, amphibians and macroinvertebrates (Gomi *et al.*, 2002). They supply nutrients, sediments, organic matter and cold water to downstream reaches. In Southern Ontario, headwater stream ecosystems are extremely important for brook charr because they provide well forested, cold, clean water and nutrients for fish to complete their lifecycles. Many headwater systems are currently being threatened in Southern Ontario by extensive land use changes, often not viewed by planners, fisheries managers, or the public, as being individually significant for resident fish species (Rosenfeld, 2002). For example, many headwater streams are not even documented on topographic maps. The lack of information on headwater streams and their resident biota presents a serious problem for stream protection and understanding the potential effects of projects, such as water extraction on stream ecosystems.

Of particular interest to this research are the many water extraction activities currently occurring in Ontario. Water extraction in the context of this project is viewed as the removal, or diversion, of water directly, or indirectly, from a stream. Direct removal relates to surface water takings, or a diversion of water within the stream. Indirect removal refers to water pumped from a well, which is hydrologically connected to the stream. If the well is hydrologically connected to the stream, the stream may respond to the pumping with a measurable reduction in streamflow or changes in other critical hydrological parameters within a definable zone of influence (Blackport *et al.*, 1995). Extractions of water from wells for commercial and municipal use are both examples of indirect water removal. Water extracted from wells can be further separated into "spring-water" and "well-water" (Blackport *et al.,* 1995). Spring-water is water that naturally flows to the surface under pressure. To collect "spring-water", collector systems are usually placed near the source of the stream. If enough water is pumped from the well, there may be a measurable response in streamflow. Well-water is water drawn from a deeper system, which does not flow to the surface under pressure (Blackport *et al.,* 1995). Well water is usually connected to either a shallow or deep aquifer (Blackport *et al.,* 1995). In either case, depending on the amount of water extracted and the connectivity of the source to surface water features, fish and fish habitat may be adversely affected by pumping activities associated with well operation.

Fish habitat potentially affected by water extraction includes all in-stream features associated with water flow required by fish to complete life cycle stages, e.g., current velocity, water depth and wetted perimeter. Habitat is particularly important in the context of water extraction because it has been hypothesized that there is a causal relationship between water extraction and changes in fish habitat and species relative abundance (Bain *et al.,* 1988; Brown and Ford, 2002; Kraft, 1972; Nuhfer, 1998; Schlosser, 1985; Stalnaker, 1981; Strevens, 1999). The nature of the causal relationship and the mechanism(s) through which it operates, however, are not well understood for most fish species. For example, a review of the literature (see below) reveals there are no long-term/high impact studies on brook charr (*Salvelinus fontinalis*) dealing with water extraction projects and their effects on brook charr populations.

Methods for Determining In-stream Flows

Since the early 1970s, there have been numerous instream flow methodologies, such as: Tennant (Montana) Method (Tennant, 1976), Flow Duration Curve Methods (Fausch *et al.,* 1984), Migration Cue Method (Annear *et al.,* 2002), Demonstration Flow Assessment (Annear *et al.,* 2002), New England Aquatic Baseflow Standard (Kulik, 1990), Biological Response to Flow Correlation Method (Conder and Annear, 1987), Single Transect Hydraulic-Based Habitat Methods (Nehring, 1979; Annear and Condor, 1984), Thompson Method (Thompson, 1972), Wetted Perimeter Method (Hamilton, 1985), Washington Toe-of-Bank Width Method (Annear *et al.,* 2002), Channel Maintenance Flows in Gravel-Bed Streams (Annear *et al.,* 2002), Flushing Flow (Empirical, Sediment Transport Modeling, and Office Based Hydrologic Models (Annear *et al.,* 2002), Median Monthly Flow (Bovee, 1982), Modified Tennant Method (Stalnaker *et al.,* 1995), Physical Habitat Simulation Model (Milhous *et al.,* 1989; Stalnaker *et al.,* 1995), Floodplain Inundation Method (Annear *et al.,* 2002), Salmon Barrier (Annear *et al.,* 2002), Tidal distribution/Estuary (Annear *et al.,* 2002), Two-Dimensional Models (Leclerc *et al.,* 1995), Feeding Station Method (Fausch, 1984; Beecher, 1987; Annear *et al.,* 2002), Riverine Community Habitat Assessment & Restoration Concept (RCHARC) Method (Nestler *et al.,* 1993), River Hydraulics and Habitat Simulation (Jowett, 1999), Hydraulic Engineering Center-6 Model (HEC-6) (Annear *et al.,* 2002), Hydraulic Engineering Centre-River Analysis System (HEC-RAS) (Annear *et al.,* 2002) and the Instream Flow Incremental Methodology (IFIM) (Stalnaker *et al.,* 1995) proposed for determining appropriate instream flows for the protection of

aquatic organisms. Many of the methods were developed in the United States, but have been subsequently adopted and modified for use in other countries (Jowett, 1997).

Instream flow methodologies can be divided into two leading groups: standard-setting (SSM) or incremental (IM) (Stalnaker *et al.,* 1995). Standard setting methods are appropriate for low-intensity decisions, where inexpensive, straightforward, rule-of-thumb solutions are required (Stalnaker *et al.,* 1995). These methods set a threshold below which water cannot be diverted for out of stream use (Stalnaker *et al.,* 1995). Standard setting methods can be further subdivided into two categories; historic flow and hydraulic methods. As the name implies, historic flow methods rely solely on the hydrologic records of a stream for setting minimum flows below which existing life forms may be impacted. The ecological goal of most historic flow methods is to sustain existing life forms by recommending a minimum flow that is within the historic flow range (Jowett, 1997). The use of stream gauge records assumes that measured flows support aquatic resources at acceptable levels (Jowett, 1997). Hydraulic methods relate parameters for describing the hydraulic geometry of stream channels to discharge, with the aim of keeping the main river channel 'full' to maximize food production (Jowett, 1997).

In contrast to standard setting methods, incremental methods are appropriate for high complexity projects, which have many decision variables (Stalnaker *et al.,* 1995). IM methods are typically more expensive, more scientifically defensible and designed for stakeholder bargaining (Stalnaker *et al.,* 1995). More commonly referred to as "Habitat Methods", IM methods are a natural extension of hydraulic methods. The difference is that the assessment of flow requirements is based on hydraulic conditions that meet specific biological requirements rather than on the hydraulic parameters themselves (Jowett, 1997).

All three categories (flow, hydraulic and habitat) aim to maintain the stream environment, but they focus on different aspects of the stream, such as flows (e.g., Tennant Method), wetted perimeter (e.g., Wetted Perimeter Method), or physical habitat (Physical Habitat Simulation Model). On a conceptual level, habitat-based methods differ from flow and hydraulic methods in that they make no a priori assumptions about the state of the natural ecosystem (Jowett, 1997). Flow and hydraulic methods assume that lower than natural flows will degrade the stream ecosystem, whereas habitat methods accept the possibility that a natural ecosystem or some

particularly valued aspect, can be enhanced by other than naturally occurring flows (Jowett, 1997). The application of each method depends on the range of conditions, such as the level of controversy or significance of the fishery (Stalnaker *et al.,* 1995).

Most of the methods currently being used by government/management agencies in other countries incorporate conservative threshold flow rules (e.g., 10% baseflow), because the scientific community cannot predict fish population sizes under alternative flow regulation scenarios (Orth, 1987). Methods are often criticized for failing to consider aspects of the stream environment (Jowett, 1997). For example, none of the methods consider temperature, water quality or biotic interactions explicitly and any change to the stream environment could potentially cause unexpected results (Jowett, 1997). As a result, Castleberry *et al.,* (1996) concluded that, "currently no scientifically defensible method exists for defining the in-stream flows needed to protect particular species of fish or aquatic ecosystems". Accordingly, more detailed comprehensive studies of stream fish and invertebrate responses to altered flow regimes will be needed before simpler and less costly methods evolve.

Water takings in Ontario are governed by the Ontario Water Resources Act (OWRA) and the Water Taking and Transfer Regulation (O. Reg. 387/04) a regulation under the Act. Section 34 of the OWRA requires anyone taking more than a total of 50,000 litres of water in a day, with some exceptions, to obtain a Permit-to-Take-Water (PTTW) from a Director appointed by the Minister for the purposes of Section 34. In evaluating an application, the Director must consider the natural functions of the ecosystem, including potential impact on the natural variability of water flow or water levels, minimum stream flow, and habitat that depends on water flow or water levels. Although the regulations do not recommend a method to determine in-stream flow requirements, they do suggest that if a qualified person can demonstrate that a proposed taking from a 3rd or higher order stream is less than 5% of the 7Q20 at the point of taking, there would normally be no need to undertake studies to establish minimum flow requirements.

In light of the above, research was begun in 2002 to investigate facets of the relationship between water extraction and the status of affected brook charr populations. The first component of the study examined linkages between flow, density and growth in age-0 brook charr inhabiting forested headwater streams in southern Ontario where groundwater was being extracted for water

bottling use. Specifically the study sought to test the hypotheses that: [1] differences in fork-lengths among sites were principally influenced by differences in stream thermal regimes as measured by accumulated thermal units, with differences in stream physical characteristics (e.g., water quality, morphology, riparian vegetation) also having significant effects; [2] among-stream differences in stream flow regimes would significantly influence among-stream differences in age-0 densities, with increases in flow being positively correlated with density; and, [3] length and density would be negatively correlated. A concurrent second study focused in more detail on the status of a brook charr population affected by municipal groundwater pumping. Objectives of the second study were to determine the effects of pumping from extraction wells proximate to the stream on stream habitat and the resident brook charr population. Specifically, the study aimed to test the hypotheses that: [1] well-related water extraction was associated with habitat degradation at sites proximate to the extraction point; and, [2] degraded habitat was associated with reduced juvenile and adult brook charr use and preference, reduced occupancy density and reduced condition.

CHAPTER 2. LITERATURE REVIEW

Water extraction has the potential to effect brook charr (*Salvelinus fontinalis*) habitat, movement, populations and temperature. These four variables are the focus of the review because the majority of research conducted to-date in respect to water extraction and its effects on fish populations have focused on these areas with the exception of spawning. Of the variables selected, most of the research has concentrated on changes in physical habitat, which is probably driven by the need for this information in stream flow models. Since the emphasis of this research is groundwater-fed headwater streams, there is a need to understand connections between physical habitat, movement, changes in population variables (e.g., density, growth) and temperature with respect to brook charr and water extraction.

Headwater streams are important habitats for fish, amphibians and macroinvertebrates. They are critical areas for nutrient dynamics and supply sediments, water, and organic matter to downstream reaches (Allan, 1995). Because of their geographical isolation, headwater systems also support genetically isolated species; thus, they support an important component of biodiversity in watersheds. For instance, new and endangered species are often found in headwater streams because such streams are relatively unexplored (Dieterich and Anderson, 2000). Despite the significant roles of headwater systems within the stream/river channel network, the ecological values of headwater systems are underestimated, and their processes have been extensively modified by land use (Meyer and Wallace, 2001).

In southern Ontario, headwater streams are extremely important to brook charr because they provide typically well forested, cold, clean water for fish to complete their lifecycles (Blackport *et al.*, 1995). Many headwater systems are currently being threatened in southern Ontario by extensive land use changes often not viewed by planners, fisheries managers, or the public as being significant for resident fish species. Agricultural and urban development pressures are extreme in the region and much of the land has been cleared of its original forests. As a result, brook charr habitats have been restricted to headwater reaches of streams where forests and wetlands remain to produce groundwater dominated habitats (Witzel and MacCrimmon, 1983; Barton *et al.*, 1985; Bowlby and Roff, 1986). Many of the remaining headwater streams are not even documented on topographic maps. The lack of information on headwater streams and their

resident biota presents a serious problem for stream protection and understanding the effects of human actions, such as water extraction, on headwater biota.

Fish habitat, includes all in-stream features required by fish to complete the life cycle. Features such as current velocity, water depth and wetted perimeter are discussed below in detail and are the focus of much of the existing literature on ecological flow needs (e.g., Poff *et al*., 1997). Habitat is important in the context of water extraction because there is thought to be a causal relationship between water removals and changes in fish habitat and relative abundance (Stalnaker, 1979). The nature of the causal relationship and the mechanism, or mechanisms, through which it operates, however, are not known with certainty (Stalnaker, 1979).

Fish movement can occur in response to physical habitat changes (Bjornn, 1971; Heggenes *et al*., 1991; Clay and Butland, 1995) and result in changes in the population structure (McFadden, 1961; Elwood and Waters, 1969; Armstrong *et al*., 1998; Covington and Hubert, 2003). In other words, as the habitat changes in response to water extraction, fish movement may be restricted and fish may move to new areas more suitable for a specific life cycle stage. The trigger to move, if not part of a natural cycle may impose a stress on the fish population, giving rise to attendant population-level stress effects. Therefore, understanding natural movement patterns for some species is very important, especially if they do not naturally move much within natal streams.

Changes in fish population variables (e.g., biomass, density, length/weight relationships, age class distributions, production, etc.) are complex and usually take several generations to respond to stresses (Power, 2002). In the context of water extraction, the time lag of a biological response is probably due to compensation mechanisms, used by fish to tolerate stressful events (Flodmark *et al*., 2002). Understanding these compensation mechanisms is extremely important, because they can mask population problems, which may be disastrous at a later date (Flodmark *et al.*, 2002; Schreck, 2000; Power, 2002).

Fish Habitat

As water is being extracted from a stream, some features of the existing habitat are likely to change, and the effect of the change will be greater in smaller streams than in larger ones (Bain

et al., 1988). Bain *et al.* (1988) noted that for any specific location in a stream, a change in stream flow translates into a change in water depth and velocity, which directly influences stream substrate and instream cover. Government agencies (e.g., Fisheries and Oceans Canada) view such influences on stream substrate and instream cover as an artificial disturbance of stream habitat stability. Furthermore, Fisheries and Oceans Canada, considers any impact on a stream that harmfully alters, disrupts or destroys fish habitat to be a violation of the *Fisheries Act* (s.35 (1)) unless authorized by the Minister of Fisheries and Oceans under section 35(2), or by regulation. Since fish species are accustomed to some degree of change in their environment, the effect on local fish populations will depend on the intensity and duration of the disturbance. Indeed, many species have evolved to tolerate, and in fact require, either torrential flows of well oxygenated water, or low flows of poorly oxygenated warm water, but not both (Cushman, 1985). Disturbances can be obvious and have large effects, or less obvious and have minimal effects, on fish populations. 'Power Ramping', as conducted by hydroelectric facilities, would be an example of a large obvious effect that disrupts habitat dramatically over very short time periods in a way that is entirely outside the natural range experienced by stream fishes, and beyond the capabilities of many species to exploit (Bain *et al.*, 1988). In contrast, water extraction for a water bottling operation can have much more subtle effects that might fall within the tolerance limits of the fish population.

Many different habitat features are required to maintain a healthy fish population. For brook charr, some habitat requirements are more important than others. Wesche (1974) identified three important habitat requirements that limited stream salmonid carrying capacity: available cover, availability of rearing or nursery areas, and presence of food-producing and spawning areas. The relative importance of habitats will change through the life cycle (Elwood and Waters, 1969). For example, optimum cover differs with age-class, with fry and fingerlings usually found in shallow riffle areas, and larger fish in deeper pools with undercut banks (Heggens and Traaen, 1988; Saunders and Smith, 1962). In small streams, it is the availability of undercut banks and pools which are considered to be the dominant factors limiting the standing crop of older brook charr (Saunders and Smith, 1955). Similarly, rearing areas shift with season and life stage. For example, brook charr winter-rearing areas consist of pools, slow reaches of runs, and stream margins, whereas summer rearing areas consist of mid-stream, riffles, or the heads of pools

(Cunjak and Power, 1986). It has been reported, however, that brook charr will use pools and slow reaches during both summer and winter (Nyman, 1970).

As brook charr develop, flow sensitivity and especially velocity, are important as the fry will stay in the substrate until the 'swim-up' stage, when the yolk sacs are absorbed (Power, 1980). After yolk absorption, fry must swim up in search of food, entering the free-feeding stage. As they develop and reach a length of 40 – 50 mm, they become able to tolerate higher water velocities (Heggenes and Traaen, 1988). In a study on critical water velocities of four salmonid species, Heggenes and Traaen (1988) reported that water temperature was positively correlated with salmonid abilities to withstand different water velocities. For example, the maximum critical velocity for brook charr was 0.17 ms^{-1} at 6-8^{o}C, 0.19 ms^{-1} at 12-14^{o}C, and 0.22 ms^{-1} at 19.2^{o}C (Heggenes and Traaen, 1988). During the first winter, age-0 brook charr will overwinter in the quieter reaches of the stream, where cover is abundant and flow is lower (Cunjak and Power, 1986). Sources of groundwater, especially sufficiently large spring-fed tributaries, are excellent areas for the overwintering of smaller fish, because temperature and flow rate are stable (Cunjak and Power, 1986). Furthermore, the small size of the headwater streams also excludes larger fish, which instead over-winter in deep main river pools (Cunjak and Power, 1986).

Flow reduction can have an effect on stream physical features such as width and depth, but neither are reduced in proportion to the reduction in the volume of flow (Wipperman, 1967). In most cases, it is velocity that is the most severely reduced parameter (Curtis, 1959; Wipperman, 1967; Kraft, 1968; Wesche, 1973; Kraft, 1972; Cushman, 1985; Bain *et al.,* 1988), followed by mean cross-sectional area (Kraft, 1972; Wesche, 1974; Cushman 1985) and mean top width and wetted perimeter (Wesche, 1974; Cushman, 1985). It is generally accepted by researchers that velocity is the driving force of stream ecosystems (Jowett, 1997). Velocity affects most of the six primary factors that influence stream ecosystems: food, physical habitat, temperature, water quality, flow regime and biotic interactions (Orth, 1987). However, channel morphology determines the eventual impact of reduced velocity. For example, Kraft (1972) reported that in response to fluctuations in discharge, the surface area, average depth and cover of runs showed a greater reduction than similar metrics in pools. Conversely, in pools, current velocity fell more than in runs as discharge fell.

Along with current velocity, numerous studies have identified water depth as another key habitat variable affecting fish distribution (Schlosser, 1982; Finger, 1982; Moyle and Baltz, 1985; Bain *et al.,* 1988). The general pattern is one of juvenile occupancy of the shallow and slow reaches, and domination of larger, older fish in the deep reaches (Bain *et al.,* 1988). However, Muller (1953) argued that mean velocity through a stream section appears to be a more significant factor in regulating brook trout populations, because invertebrate prey drift is positively related to current velocity. Chapman (1966) also found that in swifter parts of a stream, fish require less space to obtain their food, therefore territories are reduced and population densities may be higher. Wesche (1974) found that available trout cover, food-producing areas fsand spawning areas decreased most quickly when streamflow falls below approximately 25 percent annual daily flow (ADF). In the context of water extraction, where lower stream flows may cause reductions in depth and current velocity, there may be an associated indirect effect on fish populations resulting from the direct effects on benthic prey organisms.

There are numerous studies (Curtis, 1959; Wipperman, 1967; Kraft, 1972; Wesche, 1974; Bain *et al.,* 1988; Cushman, 1985; Glova and Duncan, 1985; Curry *et al.,* 1994; Nuhfer, 1998; Armstrong *et al.,* 1998; Strevens, 1999; Covington and Hubert, 2003) dealing with the effects of water extraction on physical habitat, but little has been reported about how flow reductions affect resident fish populations in a stream. In fact, many of the instream flow models used to determine threshold flows for water extraction projects use changes in habitat parameters, such as water depth or wetted perimeter, to infer a response in the fish population. Although not directly related to physical changes in flow, these methods are often criticized for failing to consider aspects of the stream environment, such as water temperature, water quality or biotic interactions (Jowett, 1997). Jowett (1997) also suggests that because of the degree of biotic diversity in rivers and the flexibility of most aquatic organisms, there is probably no sharp cut-off or single ‘minimum flow’.

Research has also been conducted using invertebrate and invertebrate habitat as potential indicators for instream flow studies. Several authors (Curtis, 1959; Hopper, 1973; Bovee, 1985) suggest that the dominant impact of fluctuating stream flows may be exerted on the food base, rather than the fish. Curtis (1959) viewed the stream bottom area as an important producer of food which varies with stream discharge. He studied the effects of water depth, water velocity,

wetted perimeter and cross-sectional area with changes in discharge by relating each parameter to the total area of stream bottom covered by water at each level of flow. Velocity of flow was found to decrease at a much faster rate than either wetted perimeter or depth. The California Department of Fish and Game developed criteria for evaluating the trout food requirements, spawning area and shelter (Stalnaker and Arnette, 1976). In 1960, the criteria were further refined and measured at four flows in Taylor Creek (summarized in Hooper, 1973). Study results showed that as flow decreased: 1) the percent loss in food producing areas and shelter were greater than the percent loss in cross-sectional and surface area; 2) the rate of loss of food producing areas and shelter increased; 3) the cross-sectional and surface area decreased at a slower rate than the volume of flow. Rader and Belish (1999) studied the impact of flow alterations on invertebrates in three small, headwater streams in the central Rocky Mountains, USA and found that invertebrates appeared resilient to diversions of ≤25% of baseflow, between November-March. However, in more heavily diverted tributaries, where the majority of flow was diverted for approximately 10-11 months, invertebrate abundance and diversity declined and spatial and temporal variation increased. Taken together these studies indicate that changes in velocity affect the population of bottom-dwellers, not only quantitatively but qualitatively, and have the potential to indirectly affect fish populations (Curtis, 1959). Since most water extraction projects in southern Ontario require the proponent to operate in a manner that does not negatively impact the natural environment (Ontario Ministry of the Environment, 1999), invertebrate studies show promise for determining appropriate ecological threshold flows for such projects.

Fish Movement

Fish need to move freely from one habitat type to another to complete different life stages. If the habitat offers adequate cover, food and spawning sites, then fish (e.g., brook charr) may not need to move at all (Shetter, 1968). Nevertheless, movement is an important element in fish survival and needs to be understood with respect to water extraction projects. The distance fish move likely depends on several factors, such as the quality of habitat available, the size of the stream and the season. For small-bodied brook charr living in a first-order tributary within southern Ontario, distance of travel can be very short, as the stream may only be 1 km long before it flows into a larger river. If water extraction reduces stream flow, current velocity and water depth (Kraft, 1968; Wesche, 1973; Cushman, 1985) fish may respond by moving away from the area of impact. For example, low flows in particular have been shown to increase activity levels

in river resident brook charr below hydroelectric facilities where ramping occurs (Murchie and Smokorowski, 2004).

Fitness optimization theory suggests that fish will migrate from shallow areas to seek refuge in pools as water levels decrease to critically low levels (Armstrong *et al.*, 1998). During drought conditions fish that remain in shallow reaches risk death as the reaches dry (Armstrong *et al.*, 1998). Fish may also move in response to a direct adverse local stimulus, such as flow reduction (Murchie and Smokorowski, 2004), and/or they might leave shallow areas if they perceive deeper ones to offer better habitat (Armstrong *et al.*, 1998). Wiens (1976) in his discussion of population responses to patchy environments, suggested that as habitat availability changes, organisms will respond by moving to better areas. Therefore, during water extraction, if flows are reduced and habitats are altered, fish should be less affected than immobile species, such as macroinvertebrates (Bovee, 1985). Assuming that fish can move and relocate to more suitable habitats. However, the fact that better habitat might be available somewhere else does not necessarily mean that fish know where that habitat is or that they will search for it. Some studies have indicated that only fish that are already familiar with available habitat options will emigrate (Armstrong *et al.*, 1998). For example, Elliot (1986) and Shirvell (1994) suggested that if salmonids are free to sample different habitat options within their home range, then they may be able to choose either deep or shallow areas within their home range as conditions change.

In respect to brook charr, there is considerable debate as to whether their home ranges are extensive or restricted. Movements of stream-resident (non-anadromous) salmonids have been studied for nearly 60 years (Gowan and Fausch, 1996). Shetter and Hazzard (1938) concluded from marking studies that during the summer months, populations of stream resident fish were relatively mobile, though within localized limits. McFadden (1961); Shetter (1968); Gowan and Fausch (1996) have found that brook charr do not restrict their activity to small home ranges within streams and that some marked fish have been observed moving at least 600 m. Curry *et al.* (2002), found that brook charr have seasonally regulated movement patterns, which includes moving into the headwater reaches during the fall and migrating downstream after spawning. In contrast, other studies (Shetter, 1937; Schuck, 1945; Stefanich, 1952; Newell, 1957; Miller, 1957; McFadden, 1961; Shetter, 1968; Solomon and Templeton, 1976; Cargill, 1980) have noted that a high proportion of recaptured fish did not move from the initial captive site. Shetter (1937),

studying brook charr in the region of Lovells, Michigan concluded that brook charr do not undertake extensive summer movements. The majority of studied fish (anywhere from 50 to 100 per cent) showed no migration (Shetter, 1937). Gerking (1958) posited that the movement of brook charr was restricted because the species is aggressive. He argued that aggression acted as a spacing mechanism (Gerling, 1958) preventing movement to adjacent habitats. Leclerc and Power (1980) also suggested that brook charr were territorial and may move very little, occupying the same sections of river throughout life. Movement may also be influenced by population size and structure (Riley *et al.,* 1992). If recruitment is too great to be supported by the habitat, or if habitat quality is lower than usual (such as in a year with low flow), territorial aggression may well have a greater impact on movement causing increased displacement from preferred habitats (Riley *et al.*, 1992).

Movement, however, may depend on dominance and fish size (Newman, 1956; Kalleberg, 1958; Chapman, 1962; Symons, 1971; Fausch, 1984; Fausch and White, 1986; Grant and Kramer, 1990). If movement depends solely on the ability to defend territories, then the largest individuals should move least. Power (1980) suggested that brook charr can be sedentary, spending their life in an area of no more than 100 m^2 or extremely mobile, migrating regularly among feeding, overwintering and spawning areas. In general, however, movement in stream and river habitat was concluded to be minimal (Power, 1980).

Unfortunately, almost all information on movement patterns comes from captures of fish at fixed locations or times during the open water or ice-free season (Curry *et al.*, 2002). Such information results in conservative estimates of true movement patterns (Gowan *et al.*, 1994). Riley *et al.* (1992) noted that determining whether brook charr move great distances or not is difficult, as most studies rely on mark-recapture methods and sampling rarely extends far enough to account for the possibility of long-range movements. In assessing recapture studies, it is important to realize that low recapture rates alone do not necessarily indicate a high degree of movement, since mortality is often high (Riley *et al.*, 1992). Recaptures of tagged brook charr on the spawning beds in the north branch of the Au Sable River, Michigan, during late October and early November indicated that slightly more than half (between 50 and 55%) of the breeding fish moved upstream a relatively small distance – on average, less than 1.5 km – to spawn (Shetter, 1937). Where habitat offered adequate cover, food and spawning sites, brook charr were

essentially sedentary (Saunders and Smith, 1962, Shetter, 1968). Curry *et al.* (2002), using radio and acoustic tags, found that brook charr moved greater distances during the spring (upstream 65–100 km) and minimal distances during the summer, where habitats consisted of deep pools and runs with cover.

Other factors that can trigger movement include water temperature (Chapman and Bjornn, 1969; Bjornn, 1971; Power, 1980; Hillman *et al.*, 1987), fish size (Miller, 1957; Solomon and Templeton, 1976), local population density (Harcup *et al.*, 1984; Hesthagen, 1988), winter conditions (Cunjak and Randall, 1993), the presence of non-salmonids (Flick and Webster, 1975), habitat characteristics (Bjornn, 1971; Hutchings, 1986; Heggenes *et al.*, 1991; Clay and Butland, 1995), food availability and distribution (Clay and Butland, 1995), reproductive sites (Clay and Butland, 1995), territoriality (Clay and Butland, 1995), predator avoidance (Clay and Butland, 1995), and flow regime (Manion, 1977; Montgomery *et al.*, 1983; Clay and Butland, 1995). Not all of these factors will apply in every stream. In particular, fish may not encounter many of these factors in small headwater streams, which are typically more stable in temperature and flow (Gowen *et al.*, 1994). Gowen *et al.* (1994) suggested that fish move less in stream systems that are more constant in temperature and flow (i.e., springs) than in harsher or more variable systems (i.e., rivers). In turn, fish that live in stable systems may have a harder time adjusting to the flow fluctuations that characterize many water extraction projects (Allan, 1995). For example, Curry *et al.* (2002) noted that brook charr movement was seasonal, peaking in summer as habitat areas dried and/or water temperatures became too high for brook charr to survive. Annual movement patterns, however, may vary, with movement in years of climate stress (i.e., droughts and floods) and decreasing in years when environmental factors are more stable (Clay and Butland, 1995).

For those studies where fish have been observed moving, there does not appear to be any obvious pattern to their movements. Movements by age-0 brook charr appear to be highly variable, but these fish generally tend to move downstream more often then do older individuals (Hunt, 1965; Phinney, 1975). Adams *et al.* (2000) in a study on the ability of naturalized brook charr to ascend steep, headwater streams in the western USA found little evidence of movements longer than 67m in either direction by marked age-0 fish. Hunt (1965) reported that the relative importance of upstream versus downstream movement by age-0 brook charr in Lawrence Creek, Wisconsin, varied by stream section but that, downstream movements predominated. Hunt (1965)

also recorded both upstream and downstream dispersal of parr movement during the first summer. O'Connor (1974) observed that the greatest downstream movement of young-of-the-year brook charr coincided with reduced volume of the streams following spring run-off and maximum population pressure, since the fish were actively establishing territories at this time. However, Gowan and Fausch (1996) found that brook charr movement was most common in the upstream direction during summer, and about equal upstream and downstream between summers. The highest rates of movement occurred during and just after runoff, and before spawning, but substantial numbers of fish moved throughout the summer (Gowan and Fausch, 1996). Similarly, Riley *et al.* (1992), reported that brook charr movement was predominately upstream.

Brook charr as a species are known to home (Gerking, 1958), whether they have been moved as a result of human activity or a natural event. Nevertheless, it is questionable how precise their homing ability is. Many salmonids, including charrs, have been shown to home to streams, lakes, or specific sites for various purposes (Arctic charr: Johnson, 1980; brook charr: Power, 1980; bull trout: Swanberg, 1997). Shetter (1937) observed that brook charr tend to remain in the initial territory of choice, leaving to spawn upstream and later returning to the same general area, probably during the winter. However, in Ganelon Creek, Québec, where 24 radio-tagged brook charr were displaced both upstream and downstream, precise homing did not occur (Belanger and Rodriguez, 2001). Most fish settled in preferred habitats (pool or glide) away from their initial site and moved very little up to the pre-spawning period (Belanger and Rodriguez, 2001). Fish that were displaced downstream showed directional upstream movement, suggesting that it was primarily olfactory cues from upstream locations, not positive rheotaxis, that mediated the return response (Belanger and Rodriguez, 2001).

Population Dynamics

The effect of water extraction on fish populations, especially brook charr, has received very little attention in the scientific literature. In fact, it would appear that more emphasis has been placed on how flow fluctuations affect physical habitat changes (Weber, 1959; Curtis, 1959) and fish movement (Manion, 1977; Montgomery *et al*., 1983; Clay and Butland, 1995), than resident fish population parameters (e.g., fecundity, density) in a stream (Kraft, 1972). A possible reason is that fish can respond quickly to short-lived, localized events, so short-term field studies fail to capture a measurable response to water extraction activities (Roghair *et al.*, 2002). One

limitation of fish population studies is that they often involve only one or a few sites within a single stream or a few geographically close streams (Nelson, 1986; Elliot, 1989; Cattaneo *et al.,* 2002). Furthermore, there is a lack of research on the relationship between hydrology and fish population dynamics over longer periods (Nehring and Anderson, 1993). Although models have been used to couple hydrology and fish populations. Fausch *et al.* (1988) reviewed models that predicted fish standing crop and concluded that most transferred poorly to other streams and questioned their ecological relevance.

Those studies that have focused on population dynamics and flow have tended to be short-term (e.g., months) and concentrated only on parameters such as fish density or abundance. Thus, it is difficult to determine whether the effects being observed were the result of a compensation mechanism or a permanent change in the population (Flodmark *et al.,* 2002). To that end, studies should be long enough and impacts should be great enough to ensure the induced responses are measurable. An important element to consider with fish population studies is that many processes affecting population dynamics (e.g., hydrological variability, food availability, and biological interactions between individuals) depend on density and others do not (e.g., temperature variation) (Elliott, 1989; Nehring and Anderson, 1993). Describing them as "compensatory," McFadden (1961) concluded that density-dependent processes helped to regulate brook charr populations in Lawrence Creek. Such processes can be related to meteorological (e.g., extraneous) phenomena and/or environmental (e.g., inherent) change, that might make one stream, or part of a stream, more productive than another (McFadden, 1961). Other researchers have found compensatory mechanisms help to control production and survival. For example, Hunt (1974) showed that homeostatic mechanisms operated to control stream wide production in Lawrence Creek, as there was consistency between obtained estimates of annual production even though there were large underlying variations in age group, year class and section estimates.

Fish density (no. m^{-2}) is one of the more studied parameters because it is commonly estimated in the field and easy to manipulate under laboratory or experimental conditions (Jobling and Reinsnes, 1986; Marchand and Boisclair, 1998). Numerous studies have shown that fish density has a negative effect on growth rates (Marchand and Boisclair, 1998). The most common explanation for the relationship is that as density increases, the relative abundance of prey and the

associated fish growth rates decrease (Henderson, 1985). Some researchers have proposed that both feeding and activity rates may increase with prey abundance for a given fish density and that feeding may decrease and activity may increase as fish density increases for a given prey abundance (Boisclair and Sirois, 1993; Madon and Culver, 1993). Others have noted that as fish density increases the number of interactions among fish (hampering, chases, aggressions) increases and that this has a negative effect on fish foraging efficiency and subsequent growth (Jobling, 1985; Boisclair and Leggett, 1989).

Groundwater extraction has the potential to affect fish density, by reducing baseflow which can lead to a reduction in the living space available to fish and alter stream morphology (i.e., reduction of riffles, runs and pools). The situation is analogous to a drought and the relationship has been well documented in such studies (Elliot, 1986; Binns, 1994; Elliot *et al*., 1997; Hakala and Hartman, 2004). Hakala and Hartman (2004) found that during the 1999 drought headwater stream morphology was altered over all streams and that brook charr densities declined significantly. Riffle area was reduced by 25 times that of pool area and the relationship between brook charr density and pool area during the drought was age dependent (Hakala and Hartman, 2004). Elliot (1987) studying population regulation in brown trout (*Salmo trutta*) in two English streams, noted that drought affected older fish more than it did younger individuals. However, fish densities fluctuate on an annual basis and short-term studies may not be able to differentiate the effects of altered flows and natural causes. As Elliott *et al*. (1997) noted in a subsequent study of draught effects on brown trout, population assessments require long-term data so that the effects of drought can be separated from natural fluctuations in population density.

Nevertheless, some short-term studies of the effects of flow reductions on brook charr densities do exist, including Kraft (1972) and Nuhfer (1998). Kraft (1972) conducted a short-term flow reduction study in Blacktail Creek, Montana to determine if a short-term loss of physical space resulted in a corresponding reduction in the brook charr population. Between June 1965 and October 1967, Kraft reduced flows by 75 and 90%, varying reductions for periods of up to three months. During flow reductions of 75% or less, no consistent changes in the number or weight of brook charr in the experimental set-ups were observed. When flows were reduced by 90%, the total number of brook charr age-I and older in the experimental runs studied was

reduced by 62%, in comparison with 20% for runs in the control sections. No consistent effect on the number of YOY fish was observed.

Nuhfer (1998) conducted a longer-term flow reduction study (1991-1997) to determine the impact of severe water extraction during the summer on a brook charr population in Hunt Creek, MI. Controlled flow reductions of 50% of summer flow (June 1 to August 31) during 1991-1994; 75% of summer flow during 1995-96 and 90% of summer flow during 1997 were implemented and fish abundance was measured in the fall after the summer flow reduction experiments. Results indicated that the abundance of YOY, yearling and older brook charr were significantly higher following reductions of 50%, 75% or 90%, in comparison to years of normal flow (1986-1990). However, abundance of the largest brook (>20 cm) was significantly lower when 90% of baseflow was diverted than during other time periods.

The results of these studies suggest that flow reductions favour YOY and yearlings, but not adult brook charr, possibly because of changes in survivorship. In contrast, other researchers (Cattaneo *et al*., 2002; Schlosser, 1985) have found that YOY and yearlings are sensitive to low flows. For example, Cattaneo *et al.* (2002), using data collected over five to eight years in 30 scattered French River reaches concluded that 0+ and 1+ brown trout were more sensitive and harmed by summer droughts, possibly because suitable habitat shrinks. The 0+ brown trout were particularly sensitive to hydrological variability, as they were not able to cope with large environmental fluctuations (Cattaneo *et al.*, 2002). Schlosser (1985) suggested that the younger age classes may be more strongly influenced by physical ("stochastic") factors such as temperature, water level and storms and that older age classes may be more strongly regulated by biotic (e.g., competition, predation) interactions. It is widely acknowledged that at high flows, trout live at greater densities and defend smaller territories, and at lower flows defend larger territories, probably as a result of relative changes in drift-feeding success (Strevens, 1999).

Water Temperature

Water temperature is a key factor in determining brook charr habitat and has been shown to be the main factor limiting the species geographic distribution (McCrimmon and Campbell, 1969). There are numerous field observations, which indicate brook charr do not usually occur in streams where water temperature rises above 20^{o}C for prolonged periods of time (Henderson,

1963). The optimal range of water temperatures for normal activity levels in brook charr is between 5° and 20°C, with 20°C often considered the incipient lethal temperature (Power, 1980). During summer periods when water temperatures in some streams increase significantly, brook charr are known to move into spring fed tributaries or to congregate near groundwater upwellings, where water temperatures are much cooler (Gibson, 1966; Cunjak and Power, 1986). Bowlby and Roff (1986) found brook charr in streams that attained 22°C, but they were located in pools where groundwater inflow decreased water temperatures by about 5°C. Cunjak *et al.* (1993) found that when water temperatures of the Little Southwest Miramichi surpassed 22°C, brook charr moved into Catamaran Brook, a groundwater fed tributary which remained about 2 – 3°C cooler than the main river.

Many headwater streams in southern Ontario are groundwater fed, which means water temperatures remain cool and constant throughout the year. The relationship of groundwater discharge to stream temperature has been well documented with groundwater approximating mean annual soil temperatures (7 - 10°C in southern Ontario) and maintaining a constant temperature within stream reaches where groundwater discharges dominate (Latta, 1965; Klein, 1979; Bilby, 1984; Cunjak and Power, 1986; Meisner *et al.,* 1988; Blackport *et al.*, 1995). Thus, changing either the amount of groundwater entering the stream or the amount of water flowing in the channel has the potential to alter stream temperature (Poole and Berman, 2001).

Although there are many factors that can affect stream temperatures, the primary determinants are climatic drivers (such as solar radiation, air temperature, and wind speed), stream morphology, groundwater influences, and riparian canopy conditions, all of which are substantially altered by various human activities (Poole and Berman, 2001). Of specific interest for discussions of groundwater extraction effects are large withdrawals from wells penetrating the phreatic groundwater network feeding a stream. If pumping from wells reduces stream flows and has a substantial influence on water temperature (Long and Nestler, 1996), there is a potential impact to brook charr populations.

Brook charr are poikilotherms and as such, water temperature is the major factor regulating their metabolic rate and energy expenditure (Gunther *et al.*, 2005). Fish growth tends to increase as temperature increases within an optimum range of temperatures (Sadler *et al.,*

1986). Brook charr age-classes have different thermal optima for growth, which may result in significantly different patterns of growth at different temperatures. McCormick *et al.* (1972) reported that the optimal range for brook charr growth was between 12 and 16^{o}C. Although it has been suggested that water temperature may be the most important environmental variable limiting fish production in lotic systems (Baltz *et al.*, 1987), Power (1966) suggested that factors other than temperature, food, space and cover limit the growth of brook charr in small streams.

Fluctuations in water temperature can also affect fish density (McRae and Diana, 2005). Since, groundwater discharges to a stream at temperatures considerably cooler than late summer air temperatures (and at temperatures warmer than midwinter air temperatures) damped temperature fluctuation occurs in streams with high groundwater influence (Blackport *et al.*, 1995). McRae and Diana (2005), while working with age-0 brook charr in the Au Sable River watershed, found mean daily water temperature fluctuation (MDWTF) was the most important correlate with fish density.

CHAPTER 3. FACTORS INFLUENCING FORK-LENGTH AND DENSITY OF AGE-0 BROOK CHARR IN HEADWATER SOUTHERN ONTARIO STREAMS

Chapter Overview

The fork-length and density of age-0 brook charr (*Salvelinus fontinalis*) can be influenced by many environmental factors, including: water temperature, morphology, riparian vegetation and flow. The objective of this study was to determine which environmental factors were most important for determining the fork-length and density of age-0 brook charr in headwater streams in southern Ontario. Differences in temperature, stream site and flow were found to significantly influence differences in age-0 length and density of brook charr among study sites, with the effect of temperature being the most significant factor in determining differences in size and density among sites. Size and density were not related. Evidence was found for the importance of physical stream variables for determining among site differences in fish size and density, with flow affecting both. The importance of temperature and flow for size, density and condition of headwater dwelling age-0 brook charr suggests that watershed developments that moderate either the thermal or flow regime of headwater streams will have negative effects on brook charr population status.

Introduction

Headwater streams are important habitats for fish, amphibians and macroinvertebrates (Allan, 1995). They supply nutrients, sediments, organic matter and cold water to downstream reaches (Allan, 1995). In southern Ontario, headwater stream reaches are extremely important for brook charr (*Salvelinus fontinalis*) because they provide typically well-forested, cold, clean water for fish to complete their life-cycles (Blackport *et al*., 1995). Agricultural and urban development pressures are extreme in southern Ontario and the forests and forest streams in which brook charr thrive have been largely removed in connection with the spread of agriculture, urbanization and other land use changes. Often such land use changes are not viewed by planners, fisheries managers, or the public as being significant for resident fish species. As a result, brook charr abundance has been reduced and habitats are now restricted to the headwater reaches of streams where forests and wetlands remain to produce required groundwater dominated habitats (Barton *et al*., 1985; Bowlby and Roff, 1986; Witzel and MacCrimmon, 1983). Many of the remaining

headwater streams in which populations survive are not adequately documented and little, if any, routine attention is paid to population status. The resulting lack of information presents a serious problem for stream protection and/or understanding the effects of increasing groundwater extraction via wells on resident brook charr populations.

Depending on stream proximity, well depth, local geology and stream size, some features of the existing stream habitats may change when groundwater is extracted The effect of the change will be greater in smaller streams than in larger ones (Bain *et al.,* 1988). For example, groundwater extraction can reduce surface flows, which may affect physical stream features such as width and depth (Wiperman, 1967). Significant changes may also be observed in mean cross-sectional area (Kraft, 1972; Wesche, 1974; Cushman 1985), mean top width and wetted perimeter (Wesche, 1974; Cushman, 1985). In most cases, flow (velocity and volume) is the most severely affected parameter (Curtis, 1959; Wipperman, 1967; Kraft, 1968; Wesche, 1973; Kraft, 1972; Cushman, 1985; Bain *et al.,* 1988), with changes likely to hold significant implications for stream ecosystems as a result of the importance of both flow characteristics as driving forces of stream ecosystems (Jowett, 1997) and habitat suitability. In particular, flow affects, directly or indirectly, all the primary factors that structure stream ecosystems, including: food, physical habitat, temperature, water quality, flow regime and biotic interactions (Orth, 1987).

Fish density is one of the more estimated measures for describing changes in fish populations, because it is amenable to field estimation and is easy to manipulate under experimental conditions (Jobling and Reinsnes, 1986; Marchand and Boisclair, 1998). Density is influenced by many factors, including: water temperature, discharge, substrate composition, and the presence of aquatic vegetation (McRae and Diana, 2005), all of which can be affected by changes in flow. Density is non-linearly related to flow, increasing from low flow to a peak before declining at high discharge (Kraft, 1972, Smith *et al*., 2006, Bjornn amd Reiser, 1991). Density, in turn, can negatively affect fish growth rates as a result of the negative relationship between density and prey abundance that can lead to reduced ration and growth (Henderson, 1985; Marchand and Boisclair, 1998; Jorgensen *et al.*, 1993; Mazur *et al.,* 1993). As fish density increases, the number of interactions (e.g., hampering, chases, aggressions) among fish may also increase and have associated negative effects on fish foraging efficiency and growth (Jobling,

1985; Boisclair and Leggett, 1989). Thus density is likely to be responsive to both the direct and indirect impacts of groundwater extraction on fish populations and habitat.

In light of possible linkages between flow, density and growth in fish, a three-year study was begun in 2002 to examine the factors influencing the fork-length and density of age-0 brook charr in forested headwater streams in southern Ontario where groundwater was being extracted for water bottling use. Because of the literature documented effects of water extraction on stream channel width, depth and wetted area, and the possible implications of the changes in those hydraulic features for fish population density, growth and survival (Hakala and Hartman, 2004), this study sought to document the habitat conditions that influence size-at-age and density of age-0 brook charr and to test the hypotheses that: [1] differences in fork-lengths among sites were principally influenced by differences in stream thermal regimes as measured by accumulated thermal units (ATUs), with differences in stream physical characteristics (e.g., water quality, morphology, riparian vegetation) also having significant effects; [2] among-stream differences in flow regimes would significantly influence among-stream differences in age-0 densities, with increases in flow being positively correlated with density; and, [3] length and density would be negatively correlated.

Methods

Study Site

This study was conducted in three first order streams in southern Ontario (Figure 3.1). Sites were selected based on the similarity of habitat features (e.g., cover, substrate), stream gradient and watershed size (Table 1). All sites were located in groundwater fed streams dominated by mixed deciduous and white cedar (*Thuja occidentalis*) cover. All sites contained monospecific populations of brook charr within study reaches. In some streams, artificial impoundments downstream contained introduced species of large-mouth bass, *Micropterus salmoides*, and rainbow trout, *Oncorhyncus mykiss*, not found in the headwater reaches. Water extraction was the only documented environmental disturbance at each of the impacted study sites (Nassaw and Crystal Springs). As far as is known, the unnamed reference stream was not impacted by any anthropogenic activity (e.g., agriculture, water extraction, damming).

The Nassaw Springs (Nassaw) [43^{o} 29' N 80^{o} 05' E] site is located in the Township of Puslinch and was used by a water bottling operation from 1999 to 2003. In the spring of 2003, the company ceased operation and water was diverted from the source and returned upstream of the study site with small (<10%) losses. The sampling site was 50 m long and located approximately 100 m downstream of the stream source. The bordering riparian vegetation consisted mainly of white cedar, with some kidney-leaved violet (*Viola renifolia*), curly horsetail (*Equisetum scirpoides*), watercress (*Nasturtium officinale*) and rough-fruited sedge (*Carex scabrata*) species located in and along the stream. The last two species are characteristic of groundwater discharge zones (Voss, 1972). A previously installed man-made barrier at the downstream end of the study stream prevented upstream migration of other fish species to the sampling site, but did not prevent fish from migrating downstream. Although poor habitat quality dominated by lack of cover and summer stream temperatures in excess of 25°C immediately downstream of the study site probably prevented much downstream migration from occurring.

The Crystal Springs (Crystal) [43^{0} 46' N 80^{o} 06' E] site is located in the Town of Caledon, formerly Caledon Township, and has been used by a water bottling operation since 1999. The stream originates in, and flows through, a white cedar (*Thuja occidentalis*) dominated coniferous swamp. The sampling site was 50 m long and located approximately 300 m downstream from the stream source at the base of the Niagara escarpment and 200 m upstream from the confluence of the Credit River. There were no barriers to fish migration within the study site.

The Reference Creek (Reference) site is located on regional (Halton) conservation property [43^{o} 29'N 80^{o} 06'E] less than 3km from the Nassaw site and approximately 35 km linear distance from the Crystal site. There were no water removal activities within the stream watershed. White cedar dominated stream bank cover, but balsam fir (*Abies balsamea*), maple species (*Acer* spp.) and yellow birch (*Betula alleghaniensis*) were present. The sampling site measured 50 m long and was located approximately 400 m downstream of the natural stream source.

Stream Habitat and Morphology

Stream channel habitats (Table 3.1) were classified on the basis of hydraulic and morphological characteristics into channel geomorphic units following Bain and Stevenson (1999). Riffles consisted of areas of shallow, turbulent water with uniform substrate composition and cross-sectional depths <0.2 m. Pools consisted of slower moving waters with a visible sorting of substrate and cross-sectional depths >0.2 m. Runs consisted of shallow, slow waters with uniform cross-sectional depths of <0.2 m.

Riparian vegetation percent cover adjacent to study sites was assessed visually. Stream substrate was classified as fines, gravels and cobble using a modified Bovee scale (Bovee, 1982), with fines including all substrate particles ≤1 mm and gravel and cobble, respectively, being in the 1-64 mm and >64 mm particle size range.

Stream channel characteristics were measured using a transect methodology (Isaak and Hubert, 2000). Transects were spaced every 5 m and wetted width was measured to the nearest centimeter. Water depth was measured at one quarter, one half, and three quarters of the wetted width transect. Habitat area (m^2) was calculated as the product of the length (m) and mean width (m) of the study reach. Habitat volume (m^3) was computed as habitat area multiplied by mean water depth (m). Stream gradient was measured for vertical drop (cm) using a clinometer PM5/360- PC (Suunto, Vantaa, Finland) over the 50m reach for study purposes.

Hydrologic Characteristics

Water velocity was measured with a Marsh-McBirney model 2100 Flo-mate (Marsh-McBirney Inc, Fredrick, Maryland, USA) electromagnetic current meter every 0.1 m across the wetted width of each transect point (Riggs, 1985). Velocity measurements were taken at 0.6 of the water depth to obtain the average velocity for each transect-point (Herschey, 1978; Riggs, 1985). Water depth was measured at each of the transect points to the nearest 0.1 cm on a meter stick immediately before the velocity measurements were made. Additional water velocity measurements were taken on a random basis during site visits. Discharge (Q) was calculated as the total volume of water flowing past the transect per second ($m^3 s^{-1}$), using the equation:

$$Q = WDV$$

Where, W = width (m), D = depth (m) and V = average velocity ($m \cdot s^{-1}$) of the transect.

Automated water level loggers (MiniTroll Std. P. and ISCO data loggers, In-Situ Inc., FT. Collins, Colorado, USA) were installed at three of the study sites to obtain continuous water depths (m). The loggers were operational from 2002 to 2004 at the Nassaw and Crystal sites, and in 2004 at the Reference site. A rating curve was developed from the discharge and continuous water level data to obtain a hydrograph from which flow ($L \cdot s^{-1}$) and velocity ($m \cdot s^{-1}$) were inferred following standard hydrological methods (e.g., Herschey, 1978). Averages were calculated from April 1 to the date of fish capture in spring, summer and fall for use in subsequent statistical analysis.

Environmental Variables

Daily precipitation data for 2002 to 2004 were obtained from Environment Canada weather stations proximate to the study sites. The Orangeville station (ID# 6155790) was located approximately 15 km of the Crystal site. The Guelph Turfgrass CS station (ID# 6143090) was located within 15 km of the Nassaw and Reference sites. Daily air temperature data from 2002 to 2004 were obtained from Onset HOBO air temperature recorders, located at each study site. The recorders were attached to trees approximately 3-5 m meters away from each stream study site. Comparisons of mean daily air temperature and total monthly precipitation were made between April and September to determine the similarity of yearly weather patterns at each study site.

Water temperature was measured at each study site every 30 minutes using in-situ Onset HOBO instream data recorders (Onset Computer Corporation, Pocasset, Massachusetts). The recorders were deployed in the middle of each study section and were set approximately 5-10 cm off the bottom of the stream to mitigate for the potential thermal effects of groundwater upwelling. The recorders were placed at each site in April 2002 and retrieved in November 2004. Data were downloaded from each site on a regular basis. Accumulated thermal units (ATU) were calculated for each site by summing daily mean water temperatures $\geq 0^{o}C$ from April 1st, the assumed date of fry emergence (McFadden, 1961; Power, 1980) to the date of capture. Where data were missing due to recorder malfunction (approximately 2% of all data), observations were

interpolated using measured site-specific temperature data and water/air temperature regressions estimated following procedures outlined in Power and Van Den Heuvel (1999).

Fish Capture

At each site, a 50 m section of stream was isolated with blocking nets and a Smith-Root 12B-POW (Smith-Root Inc., Vancouver, Washington, USA) backpack electro-fishing unit was used to capture fish. The depletion method (Zippin, 1958) was used to estimate the abundance of age-0 brook charr in the spring (May-June) of 2003 and 2004, summer (July) of 2002-2004 and the fall (September-October) of 2003 and 2004. Electrofishing effort was standardized by thoroughly searching all stream habitats during each pass and having the same person run the electrofisher, accompanied by one netter. Approximately equal effort, as measured by electrofishing seconds, was used on each pass. Sampling minimized electro-fishing and temperature-induced changes in fish behaviour that might violate the assumption of constant catchability employed by closed-population removal estimators (Zippin, 1958) by leaving reaches undisturbed for a minimum of 30min between electro-fishing passes and electro-fishing only when water temperatures fell in the range of normal activity levels for brook charr (Power, 1980). The electro-fisher was operated with pulsed direct current at 60 pulses/s and on a 10% duty cycle. Peak voltage was generally set at 350V, but was adjusted (range=325-400 V) to maintain fishing effectiveness according to the mean width, depth, and conductivity of each study stream.

Captured fish were measured to the nearest 1.0 mm (fork-length, FL) and weighed to the nearest 0.1 g. Literature-based length-frequency histograms were used to separate fish into age-0 and older age categories (e.g., McCrimmon, 1960, McFadden, 1961, McFadden *et al*., 1967, Power, 1980, Power and Power, 1995). In the Fall of 2003 and 2004, scales were removed from a subsample of captured fish to validate the length-frequency age-group separation. Linear regression was used to test for significant linear relationships between length-frequency assigned and scale determined ages and to determine whether the regression intercept and slope, respectively, deviated significantly from zero and unity (Zar, 1999). Age-0 length data were subsequently used to estimate mortality (M) based on median length (Hoenig *et al.*, 1983) as:

$$Z = 0.693 / (Y_{MEDIAN} - Y_X)$$

where Z is the instantaneous rate of mortality ($M=1-e^{-Z}$), $Y_{MEDIAN} = -\ln(1- L_{MEDIAN} / L_{\infty})$, L_{MEDIAN} is the median length of fish captured above length L_X, the length at which fish are equally vulnerable to the capture gear (Miranda and Bettoli, 2007), $Y_X = -\ln(1- L_X / L_{\infty})$ and L_{∞} is the asymptotic length of fish as estimated from an appropriate von Bertalanffy analysis (see below). Significant differences among estimates were established by construction of 95% confidence intervals using the approximate variance for Z as described in Hoenig *et al*. (1983).

Population Density Estimates

The numbers of fish captured on each electro-fishing pass were used to estimate stream population abundance using the three pass maximum likelihood estimator of Junge and Libosvársky (1965).

$$N_t = \frac{6X^2 - 3XY - Y^2 + Y\,(Y^2 + 6XY - 3X^2)^{0.5}}{18\,(X - Y)}$$

Where, N_t is the population at time t, $X = 2y_1 + y_2$, $Y = y_1 + y_2 + y_3$ and y_1, y_2 , y_3 are respectively, the number of fish captured in the first, second and third samples. Abundance estimates were divided by stream area to obtain density estimates (e.g., numbers of fish per 100 m^2) to standardize for differences in sample site areas. The constant probability of capture assumption of the Zippin method was tested using a chi-square based statistic as described in Seber (1982).

Statistical Analyses

All statistical analysis were performed using the statistical program STATISTICA (version 8.0). Significance in all statistical procedures was judged at the $\alpha = 0.05$ level. The statistical significance of any differences among stream morphology, hydrology and environmental variables was determined with ANOVA followed by Tukey's HSD post-hoc test, after establishing variance homogeneity (e.g., Levene's test). Differences in the age-0 biological variables fork-length, weight and density were similarly tested. Linear regression was used to estimate weight-length relationships (Wootton, 1998) for the data obtained each year at each study site and regressions were assessed for statistical adequacy using standardized error term

tests (Zar, 1999). The prevalence of isometric or allometric growth was assessed by testing for significant differences between the estimated weight-length regression slope coefficient and the theoretical value of three that indicates isometric growth (Wootton, 1998). Differences in the estimated weight-length relationships by year at each study site, and among study sites were tested using ANCOVA (Zar, 1999).

Length-at-age (mm) as a function of the number of calendar days between April 1 and date of capture data were used to estimate standard von Bertalanffy growth models (Wootton, 1998) for each site. Among site model differences were examined using analysis of residual sum of squares (Chen *et al*., 1992; Haddon, 2001) as follows:

$$F = \frac{(RSS_P - \sum_{i=1}^{m} RSS_i) \Big/ (DF - \sum_{i=1}^{m} DF_i)}{\sum_{i=1}^{m} RSS_i \Big/ \sum_{i=1}^{m} DF_i}$$

Where, RSS_p defines the residual sum of squares derived from the von Bertalanffy curve estimated using the age-length data from all compared populations (m), RRS_i is the residual sum of squares from the ith compared population von Bertalanffy model and DFp and DF_I, respectively, are the associated degrees of freedom for the pooled and ith population-specific growth models. (Haddon, 2001).

The hypothesis concerning the relationship between fork-length and ATUs was tested using univariate regression with data for each study site. Differences between study sites were tested with ANCOVA (Zar, 1999). The importance of stream physical characteristics for fork-length was established with multivariate regression models using forward selection in a stepwise regression procedure (Draper and Smith, 1981). An α=0.05 threshold was used to determine the required F-to-enter and F-to-remove values (Draper and Smith, 1981), with sensitivity tests conducted on the resulting variable selection by lowering and raising the F-to-enter and remove criteria (Draper and Smith, 1981). The candidate set of variables included in the stepwise regression procedure included a classification variable (Agresti, 2002) for study site (1=Reference, 2=Crystal, 3=Nassaw) to account for differences in unmeasured parameters (e.g.,

benthic secondary production), year, area, ATU, flow, velocity, cover, depth, watershed area and wetted width. Standardized regression coefficients were computed and used to judge the relative influence of included explanatory variables on the dependent model variable (e.g., Cox, 1987). A secondary test of the importance of ATU for differentiating among populations was run by regressing site weight-length regression slope coefficients against ATUs and the set of other candidate stream morphology and environmental variables in a forward selection stepwise regression procedure as described above.

The hypothesis concerning age-0 densities was similarly tested with univariate regression models relating flow to age-0 density and multivariate regression models with variable selection driven by forward selection in a stepwise regression procedure as described above. Finally, the hypothesis concerning the relationship between length and density was tested with univariate regression.

Results

Stream morphology, hydrology and environmental variables varied among the study sites (Table 3.2), with ANOVA indicating significant differences in wetted widths and depths (Tukey's HSD $P<0.05$) but no differences in discharge, velocity or stream temperatures (Tukey's HSD $P>0.05$). Some differences between years at study sites were evident, but no consistent pattern of among year differences was observed (Table 3.2). Total monthly precipitation did not differ significantly between the Environment Canada stations at Orangeville and Guelph, during the study period 2002 to 2004 ($F_{5,65}=1.17$, $P=0.33$).

A total of 520 age-0 brook charr were captured during the spring, summer and fall samplings of 2002–2004. Length-frequency assigned and scale determined ages compared favourably ($r=0.968$, $P<0.001$), with regression of length-frequency assigned ages on scale determined ages yielding a regression with zero intercept ($P=0.353$) and a slope (0.968) not significantly different than unity ($P=0.368$). Sampling site mean fork-lengths, weights and associated standard errors for all captured age-0 fish are given in Table 3.3. Nassaw fish were longer and heavier, than Crystal or Reference fish in spring and summer (Tukey's HSD $P<0.05$). No significant differences were observed between Nassaw and Crystal fish in the fall (Tukey's

HSD P>0.05). There were significant among year differences in fall mean fork-lengths at the Crystal ($F_{1,67}$=79.84, P=0.015) and Nassaw ($F_{1,35}$=74.05, P=0.012) study sites, but not at the Reference ($F_{1,44}$<0.01, P=0.973) site. At Crystal fork-length was largest (62.2 mm) in 2004 and smallest (56.3 mm) in 2003, while at Nassaw the reverse was true (58.6 mm in 2004 and 66.3 mm in 2003).

Mean age-0 fork-length at capture and ATUs were significantly correlated at each study site (Table 3.4), with ATUs explaining ≥94% of observed variation in length. ANCOVA indicated no significant differences among the slopes of the site-specific ATU regressions (F=0.301, P=0.745), but intercepts differed (F=9.855, P=0.002), with the Nassaw site intercept being significantly higher than the estimates for either the Crystal or Reference sites. Stream morphology variables, e.g., channel width and depth, did not significantly improve on the ATU-based regression model for all sites. Classification variables for stream site and flow did improve overall explanatory power from 0.905 to 0.975, with regression ANOVA yielding P≤0.012 associated with the sequential addition of each variable (Figure 3.2). Standardized regression coefficients indicated ATUs were the most significant determinant of length, being between four and seven fold as important for determining variability in length as either flow or status.

With one exception estimated weight-length relationships evidenced isometric growth at all sites for all years, with statistical tests of the difference between the estimated weight-length slope coefficient (range: 2.24–3.42) and the theoretical value of three establishing the equivalence of the estimated weight-length slope coefficient to three (P≥0.114). The single exception was growth in 2003 at the Reference Site where allometric growth prevailed (P=0.010). There was a significant seasonal effect ($F_{2,16}$=8.212, P=0.004), with spring and fall estimated average slope coefficients differing significantly from one another (Tukey's HSD, P=0.016) and a general tendency for the length-weight coefficient to decrease from spring to fall. Among years, ANCOVA indicated no significance differences and the applicability of a common slope weight-length model for all data at Crystal Springs ($F_{2,201}$=0.46, P=0.632), Nassaw ($F_{2,216}$=0.48, P=0.620) and the Reference site ($F_{2,85}$=1.82, P=0.168). When data for all years were grouped by site, ANCOVA also indicated no significant differences in the slopes of the weight-length regression models among sites ($F_{2,520}$=1.03, P=0.357), but heterogeneity of the intercepts ($F_{2,522}$=10.06,

P<0.001). When modelled as a function of physical stream variables, estimated length-weight coefficients varied significantly only with ATU ($F_{2,17}$=18.294, P=0.001, r^2=0.518, Figure 3.3).

Sufficient length-at-age data were collected in 2003 and 2004 to estimate von Bertalanffy growth models for all sites (Table 3.5). Analysis of residual sum of squares indicated there were no significant among site differences in the models ($F_{4,14}$=1.78, P=0.189) such that the common model (Figure 3.4) could be adequately used to describe length-at-age in all sites.

The density of age-0 brook charr did not differ significantly among sites in any season (Spring: $F_{1,2}$=15.16, P=0.06, Summer: $F_{2,6}$=3.29, P=0.109, Fall: $F_{2,3}$=0.70, P=0.561). Significant differences were observed between years ($F_{2,16}$=4.24, P=0.033). Densities were generally low, averaging, respectively, at the Nassaw, Crystal and Reference sites 0.49, 0.29 and 0.15 individuals m^{-2}. Although density was not related to mean length ($F_{2,17}$=0.59, P=0.452), density could be explained by a multivariate regression model including a site classification variable, year and flow as explanatory variables (r^2=0.612, all coefficient P≤0.046). Standardized regression coefficients (β) indicated all variables were of approximately equal importance for determining variability in density, with β values for site, year and flow, respectively equalling 0.547, 0.546 and 0.353.

Fall mortality estimates (Figure 3.5) varied from a high of 0.995 at Crystal in 2003 to a low of 0.939 at Nassaw in 2003, but did not differ significantly among sites ($F_{2,3}$=0.04, P=0.963). Similarly, in summer mortality rates varied from a high 0.999 at the Reference site in 2002 to a low of 0.972 at Crystal in 2004, but did not differ significantly among sites ($F_{2,6}$=0.60, P=0.581).

Discussion

Differences in temperature, stream site and flow influenced differences in age-0 length and density of brook charr in study headwater streams. The effect of temperature, as measured by accumulated thermal units, dominated other influences, providing strong support for the study null hypothesis that size differences among sites were primarily driven by differences in the prevailing thermal regime. Evidence was also found for the affect of physical variables on differences in fish size and density, with flow affecting both to a moderate degree. Accordingly,

there was support for the secondary hypothesis regarding the importance of differences in stream physical characteristics for understanding variation in age-0 length and density among study sites.

Flow had a pervasive influence on age-0 length and density, accounting, respectively, for a 15 and 35% change in the variability of length and density for a unit change in flow variability. Although statistically significant, flow affects were dominated by other physical and biological features of the study sites as captured in the site classification variable (e.g., stream morphology, degree of groundwater influx, benthic secondary production). Thus while results provide support for the hypothesis concerning flow-related impacts on age-0 brook charr in headwater streams, changes in flow cannot be considered of prime importance over the range of stream discharge differences (14.3-28.5 $L \cdot s^{-1}$) observed here. Finally, there was no support for the third hypothesis concerning the negative correlation between age-0 length and density.

Water temperature has been described as the most important environmental variable limiting fish production in lotic systems (Baltz *et al*., 1987) and is known to critically influence brook charr growth (Power, 1980). The dominant effect of temperature on age-0 length found here relates directly to the differences between the prevailing thermal regime and the range of temperatures (12-16ºC) over which brook charr growth is optimized (Fry *et al.*, 1946; Davis, 1956; Haskell *et al.*, 1956; Patrick and Graf, 1962; McCormick *et al.,* 1972; Sadler *et al.,* 1986). In groundwater dependent streams that have temperatures below the optimum for growth, flow augmented surface area (e.g., wetted width) and radiant heat exchange with air probably serve as key mechanisms for increasing water temperatures toward the optimum range for growth. Generally, water temperatures increase as solar radiation and air temperature increase, and as water velocity decreases (Theurer, 1982). The overall affect of radiant exchange, however, appears dampened by available groundwater inflows insofar as daily fluctuations in temperature (e.g., mean daily water temperature fluctuation) had no affect on age-0 densities as reported elsewhere (e.g., McCrae and Diana, 2005).

Temperature alone will not permit increased growth unless matched by increases in ration because of the implications of higher temperatures for metabolic demands (Elliott, 1994). Temperature, however, is also a critical determinant of benthic secondary production, doubling species' growth rates for each 10°C increment within the limits of thermal tolerance (Allan and

Castillo, 2007). The positive correlation between ATU and capture size, therefore, suggests sufficient matching of temperature-driven increases in metabolic demand and ration, even in streams marginally affected by water abstraction.

Although increases in size are observed, increases in condition are not, as indicated by the negative correlation between measured weight-length regression slopes and ATU. The correlation of season with ATU (r=0.973) suggests a seasonal mismatch between ration and somatic needs that does not permit coincident growth in length and weight as fall approaches. Late season growth in the studied streams appears to emphasize gains in length over weight, probably because of the implications of length for overwinter survival in salmonids (e.g., Hunt, 1969; Metcalfe, 1998).

At regional scales fish distribution has been related to climatic, geographical, and hydrological factors, whereas at local scales fish abundance is more influenced by habitat factors (Jowett, 1990). For example, for brown trout (*Salmo trutta*) in New Zealand streams and rivers, flow mediated variations in water velocity are thought to be the most important hydraulic variables because of their linkages to changes in water temperature, benthic invertebrate and periphyton community structure (Jowett and Duncan, 1990). Similarly, in English chalk streams positive correlations between mean April discharge and September age-0 brown trout density have been documented, with the effect of flow thought to operate through an increase in suitable stream area for establishing territory or via the flow associated increase in velocity that reduces defendable territory size (Solomon and Paterson, 1980). Thus variations in flow ought to correlate with abundance and/or density, with both increasing as flow increases available instream habitat (Jowett *et al*., 2005).

In hydrologically stable environments, the relationship between density and flow may weaken, with flow negatively affecting density when increases coincide with key life-history events such as fry emergence as has been described for brook charr in Michigan streams (Zorn and Nuhfer, 2007a). Negative effects of high flows on year-class strength have been documented elsewhere in Michigan streams (e.g., Nuhfer *et al*., 1994), but overall the hydrologic stability and low-gradient nature of most Michigan brook charr streams is believed to interrupt the linkages between flow and flow-induced variation in brook charr populations (Zorn and Nuhfer, 2007a).

Earlier studies on the Pigeon River, Michigan, linked increased abundance of age-0 brook charr to high groundwater levels, with higher groundwater levels increasing stream carrying capacity during the critical post-emergence period in spring (Latta, 1965).

Although the range of flow changes observed here (14.3-28.5 $L \cdot s^{-1}$, CofV=10.8%), was small, flow was positively correlated with the density. In groundwater dependent systems, such as those studied here, increases in groundwater level will be directly reflected in changes in flow (Freeze and Cherry, 1979) that are likely to influence critical post-emergence survival as a result of increasing the availability of suitable habitat. The persistence of the flow-related effect, however, will also depend on food availability, with flow influencing food availability through increases in drift rate (Grant and Noakes, 1987; Allan and Costillo, 20007). At low flows, increased stream flow will increase total habitat area, decreases effective conspecific density and competition and increase drifting invertebrate prey delivery rates (Nislow *et al*., 1998), providing both increased ration and decreased competitive costs likely to be reflected in improved survival and density as observed here. Among study differences in the reported importance of flow likely relate to variability in site-specific scale factors (e.g., gradient, cover, channel morphology) not included in each of the studies or the effect of community composition and inter-specific competition that reduces the relative importance of flow as a determinant of brook charr success.

Many studies have investigated the effects of salmonid population density on growth, mortality, or emigration rate (e.g., Egglishaw and Shackley, 1977; Mortensen, 1977; Elliott, 1984) or tested the general idea that space limits salmonid density (e.g., Chapman, 1962; Mason, 1969). Experimental manipulations of age-0 brook charr densities have shown negative effects on growth in the laboratory (Marchand and Boisclair, 1998), although field cage experiments have provided contrary evidence (Zimmerman and Vondracek, 2006). The inverse relationship is usually explained by reductions in the quantity or quality of prey consumed by individual fish, under the assumption that food is limiting. The lack of a relationship in this study may relate to the availability of excess ration in the streams resulting from the low densities observed in all streams. Zippin population estimates when combined with the length - territory size relationship reported in Grant and Kramer (1990) suggested at most 12.5% of the available habitat could be accounted for by required territory sizes. Alternatively, recent theoretical work has suggested population growth rates are similar across a broad range of abundances (densities) because of the

concave nature of the relationship between the two (Sibly *et al*., 2005). In stable environments, where populations may be at, or near carrying capacity, fluctuations in abundance (density), therefore, are unlikely to manifest themselves in either measurable relations between density and mortality or density and growth (size).

Lower than average densities (e.g., Grant and Kramer, 1990), may be related to high mortality, ranging from 0.94 to 0.99 over the egg to October period. While studies of brook charr population dynamics point to high mortalities during the first months of life (Latta, 1962; Le Cren, 1973; Elliot, 1985; Power, 1980) and many studies have found high winter mortality (Hunt, 1969; Cunjak and Therrien., 1998; Hutchings *et al.*, 1999), survival in other seasons can be dramatically influenced by episodic events, such as droughts and floods (Carlson and Letcher, 2003). Thus estimated rates vary among studies. For example, Smith (1947) estimated mortality for brook charr from egg to 4½ month fingerlings to be 84%. Copper (1953) reported mortality rates from egg to nine month fingerlings of 97.3%. Similarly, McFadden (1961) reported nine month mortality rates in Lawrence Creek, Michigan between 1953 and 1956 averaged 98%. Latterly, Zorn and Nuhfer (2007) have reported year over year age-0 mortality rates in the range of 74 to 81% in the Au Sable River, Michigan.

The importance of temperature and flow for size, density and condition of headwater dwelling age-0 brook charr suggests development actions moderating either the thermal or flow regime of streams will have negative effects on population status. Among those activities, water extraction risks altering habitat in unfavourable ways by interrupting important linkages between flow and temperature that favour growth and density but that prevent critical physiological thresholds for temperature from being surpassed. Caution is particularly necessary given the low densities at which headwater resident populations appear to exist.

TABLES

Table 3.1. Study site mean habitat features, based on April 1 to September 15, 2002-2004 measurements (n≥10). Substrate bed materials were classified to size following a modified Bovee (1982) scale. C of V % defines the among site coefficient of variation in percent terms for each habitat feature.

Sites	Watershed Size (ha)	Gradient (%)	Riparian Vegetation % cover	Substrates			Channel Geomorphic Units		
				% Fines	% Gravels	% Cobble	% Riffle	% Run	% Pool
Nassaw	25	< 2	80	60	30	10	40	60	0
Crystal	100	< 2	50	60	30	10	30	60	10
Reference	50	< 2	80	50	40	10	30	60	10
C of V %	65.4	0.0	24.7	10.2	17.3	0.0	17.3	0.0	86.6

Table 3.2. Mean stream morphology, hydrology and environmental values measured at study sites, April 1 to September 15, 2002 - 2004. Among site differences in means are categorized with alphabetic superscripts, with means that are not significantly different sharing a common superscript. Among year differences for each study site are categorized with numeric superscripts, with superscripts indicating patterns of yearly differences as follows: 1 – 2002 and 2003 differ from 2004, 2 – 2003 and 2004 differ from 2002, 3 – 2002 and 2004 differ from 2003, and 4 – 2002 and 2004 and 2002 and 2003 form groups with common means. C of V % defines the among site coefficient of variation in percent terms for each habitat feature.

Sample Sites	Wetted Width (m)	Mean Water Depth (m)	Discharge (L/s)	Velocity (m/s)	Water Temperature (^{o}C)
Nassaw	1.53 ^{A}	0.13 ^{A}	20.9 ^{A}	0.13 ^{A}	8.8 ^{A}
Crystal	2.19 ^{B,1}	0.15 ^{B,2}	25.1 ^{A,3}	0.11 ^{A}	8.2 ^{A}
Reference	2.51 ^{B}	0.16 ^{A,B,4}	25.6 ^{A}	0.10 ^{A,4}	9.3 ^{A}
C of V %	24.1	10.4	10.8	13.5	6.2

Table 3.3. Mean fork-length (mm), weight (g), standard error (SE) and sample size (n) of age-0 brook charr captured at study sites in the spring, summer and fall, 2002-2004. Within each column, superscripts denote like means as determined using Tukey's post-hoc HSD test.

Site	Number Captured			Fork-Length±Std Error (mm)			Weight±Std Error (g)		
	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Nassaw	42	143	37	39.1±0.8 ^{A}	49.8±0.6 ^{A}	63.4±1.5 ^{A}	0.79±0.05 ^{A}	1.57±0.06 ^{A}	3.34±0.23 ^{A}
Crystal	21	118	69	33.0±1.2 ^{B}	46.5±0.7 ^{B}	60.5±1.1 ^{A,B}	0.47±0.06 ^{B}	1.19±0.05 ^{B}	2.77±0.15 ^{A,B}
Reference		44	46		43.9±0.9 ^{B}	58.0±1.4 ^{B}		1.07±0.07 ^{B}	2.61±0.17 ^{B}

Table 3.4. Site-specific fork-length-ATU regression models and the common slope model for all age-0 brook charr, 2002-2004. Site is an index variable defining site: 1= Reference, 2= Crystal and 3 = Nassaw. β is the absolute value of the standardized regression coefficient defining the relative importance of an independent variable in a multiple regression.

Site	Coefficient	Coefficient Estimate	P-value	r^2	β
Nassaw	Intercept	29.335	< 0.001	0.981	
	Slope	0.020	< 0.001		
Crystal	Intercept	24.899	< 0.001	0.941	
	Slope	0.021	< 0.001		
Reference	Intercept	25.506	< 0.001	0.963	
	Slope	0.019	< 0.001		
Combined	Intercept	215.45	< 0.001	0.975	
	ATU	0.020	< 0.001		0.997
	Site	- 5.053	< 0.001		0.245
	Flow	0.307	0.002		0.148

Table 3.5. Site-specific von Berlanffy growth models and the common model for all age-0 brook charr, 2002-2004.

Site	Coefficient	Coefficient Estimate	P-values	r^2
Nassaw	L_∞	72.158	< 0.001	0.985
	K	4.024	0.002	
	t_0	0.080	0.001	
Crystal	L_∞	71.605	0.002	0.955
	K	3.474	0.035	
	t_0	0.079	0.015	
Reference	L_∞	68.612	0.001	0.978
	K	3.545	0.024	
	t_0	0.070	0.011	
Combined	L_∞	71.306	< 0.001	0.929
	K	3.557	< 0.001	
	t_0	0.071	0.001	

Figure 3.1. Location of study sites in southern Ontario.

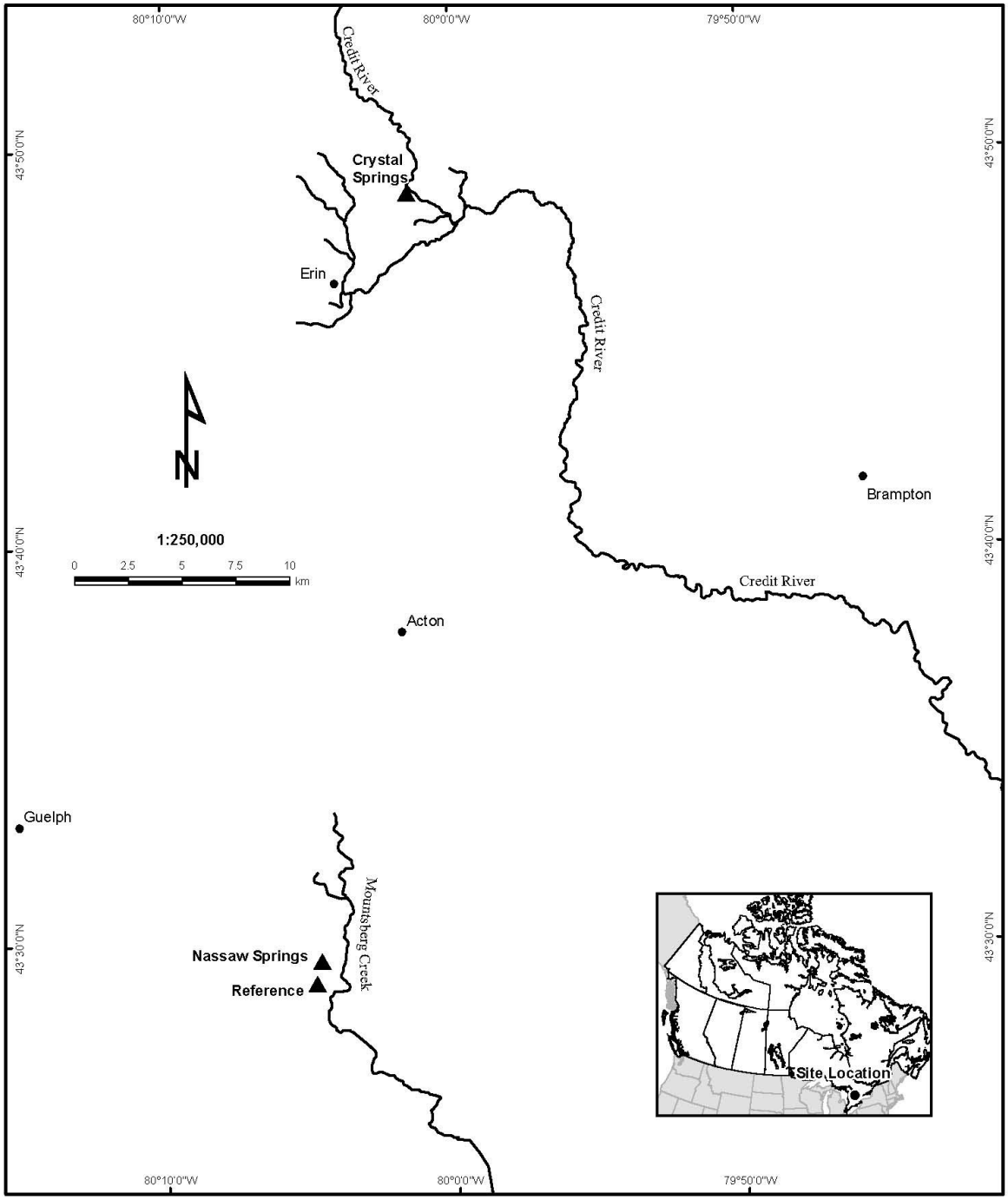


Figure 3.2. Observed and predicted fork-length of age-0 brook charr and associated 95% confidence intervals. Predictions obtained from the multivariate regression models relating ATU, site status and flow to observed age-0 length.

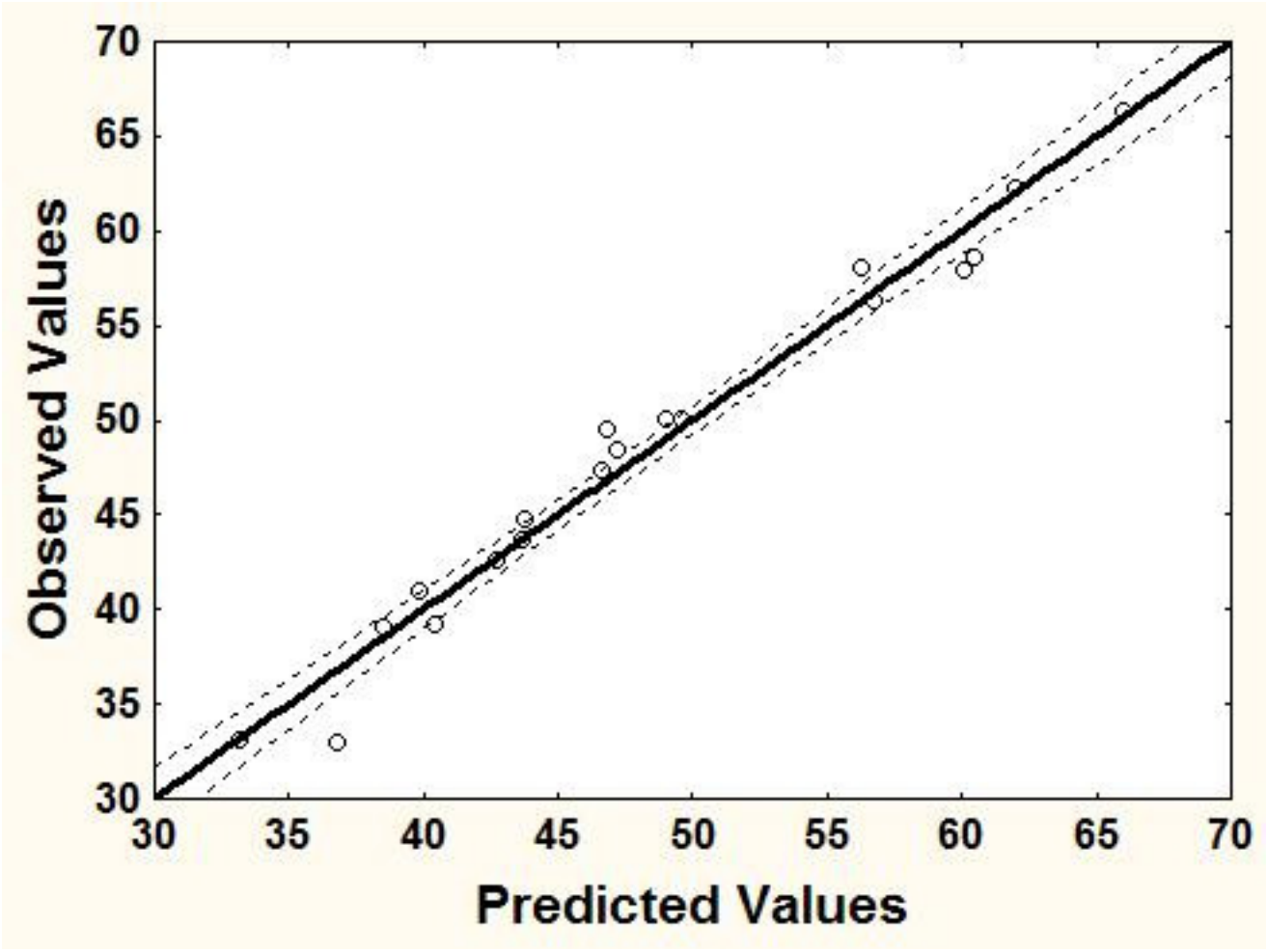


Figure 3.3. Plot of site weight-length regression slope coefficients (black circle) obtained from seasonal samples against ATU. The solid line plots the regression model relating the weight-length slope coefficient to ATU. The dashed lines plot the associated 95% prediction confidence intervals.

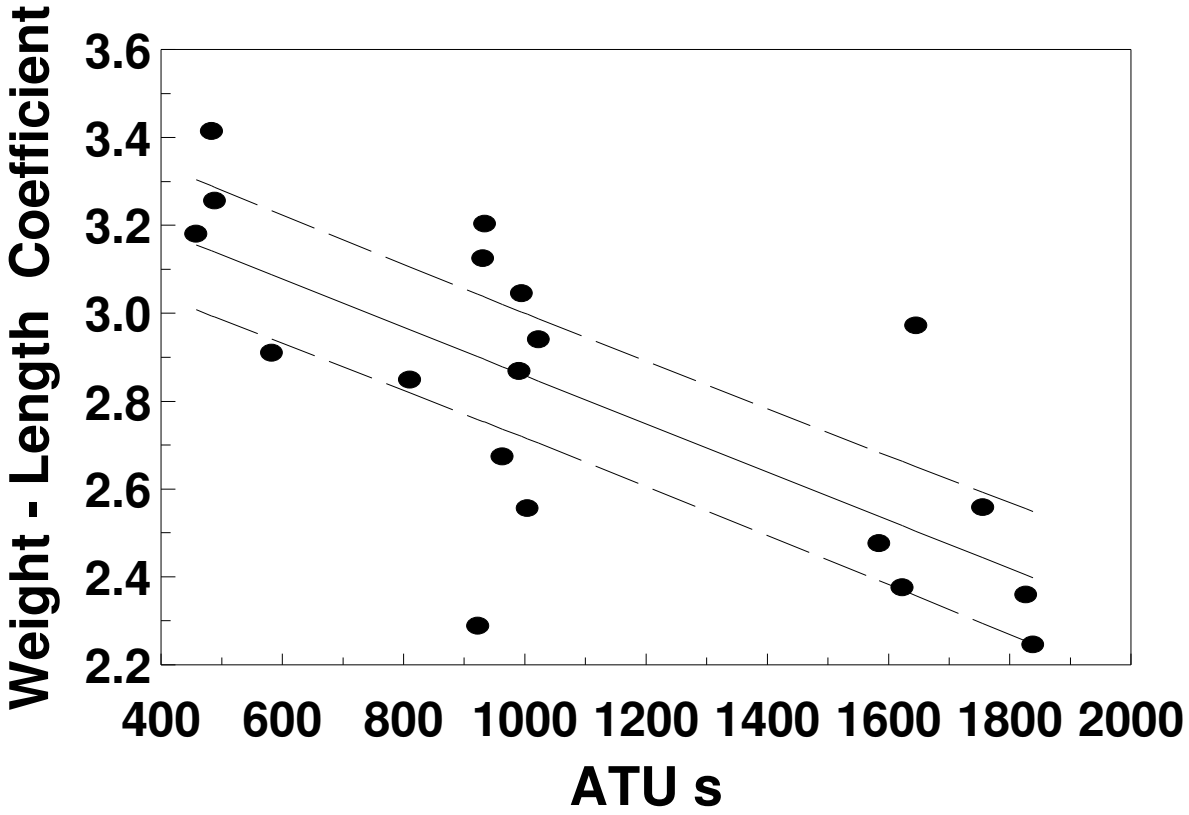


Figure 3.4. Site-specific mean length-at-age ± standard error of sampled fish and estimated von Bertalanffy growth model (solid line). Data for Nassaw, Crystal and the Reference sites, respectively, are plotted as gray, black and white circles. Standard errors of the means are plotted as vertical bars.

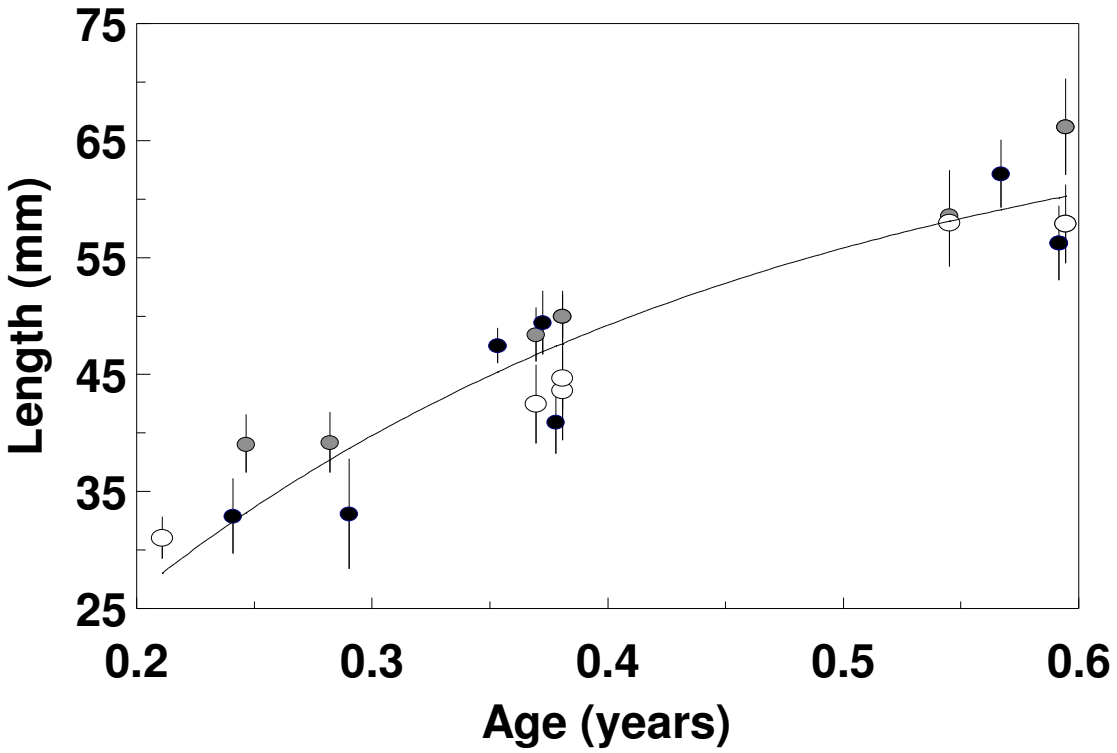
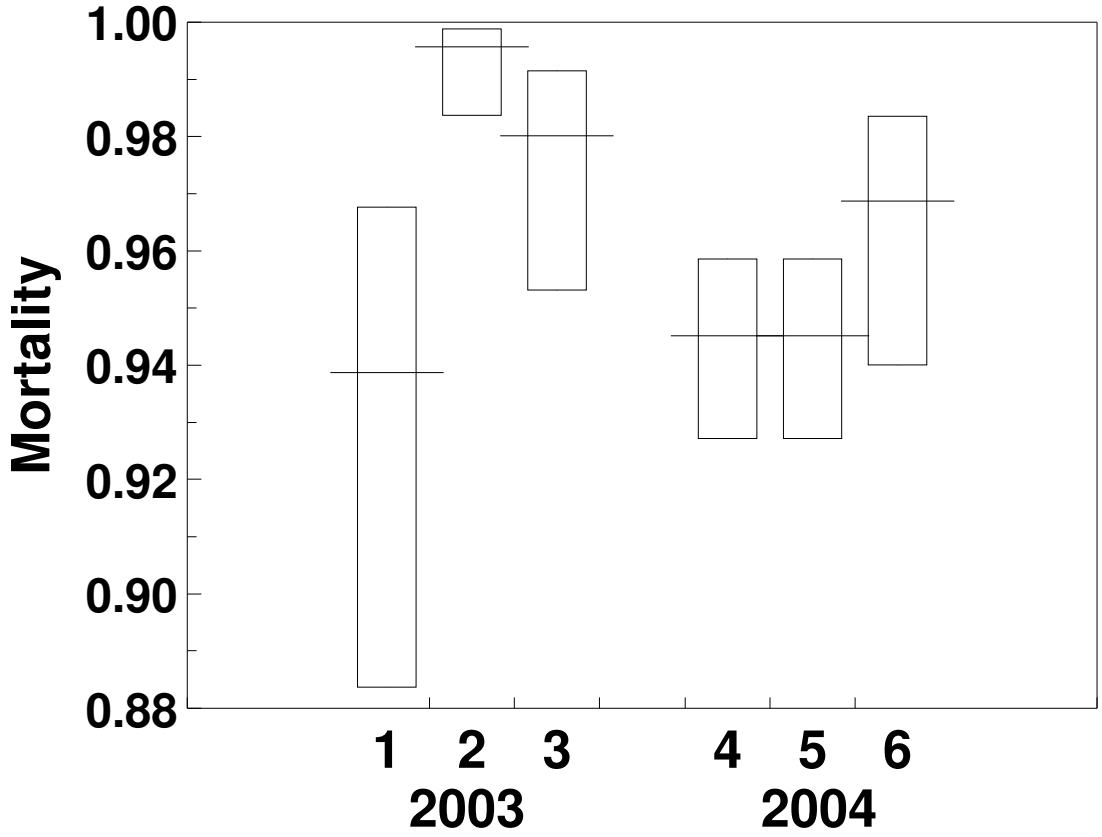


Figure 3.5. Fall mortality rate estimates for age-0 brook charr at sample study sites in 2003 and 2004. Mean values are plotted as horizontal lines within each box. Boxes define mean ± 95% confidence intervals for the estimate. There were no significant differences in estimated mortality rates among sites. Mortality rates for 2003 at Nassaw, Crystal and Reference sites, respectively, are represented by boxes 1 through 3. Mortality rates for 2004 at Nassaw, Crystal and Reference sites are similarly represented by boxes 4 through 6.



CHAPTER 4. THE EFFECTS OF GROUNDWATER EXTRACTION ON STREAM RESIDENT BROOK CHARR IN MONORA CREEK, ONTARIO

Chapter Overview

Groundwater extraction is thought to have a significant influence on the habitat, reproductive success, density and movements of brook charr (*Salvelinus fontinalis*). Here the effects of pumping from municipal groundwater wells operated by the Town of Orangeville on the stream habitat and resident brook charr populations of nearby Monora Creek are studied. Results indicate significant, although subtle, changes in available habitat as a result of pumping activity, with habitats in the Impact Site showing significant decreases in hydraulic gradient, stream depth and the availability of under-cut bank and run and pool habitat. Combined, the changes in habitat influenced use, with age-0 densities declining in the Impact Site relative to the non-impacted Sites A and B. When adjusting for relative availability, standardized preference indices indicated a significant preference for non-impacted Site A among age-0 brook charr. Adults evidenced slight changes in distribution among the available sites, with reduced densities in the impacted site and preference for non-impacted habitat in Site B in the fall. Changes in condition (weight-length relationship) and mortality were generally not in evidence during the course of this study. Thus overall, the impacts on brook charr populations subjected to moderating pumping appear to relate more to patterns of within stream distribution and habitat use than to any direct measures of fitness such as survival or growth.

Introduction

Brook charr (*Savelinus fontinalis*) occupy a wide range of ecosystem types, including small headwater streams and lakes in the southern portion of the distributional range through to large rivers and lakes in the northern portion of the distributional range (Power, 1980). Within those ecosystem types brook charr may evidence a variety of life-history tactics, including anadromy, and will vary in size, growth, and maturation as a function of the critical ecological drivers (e.g., temperature, productivity, competition) of the ecosystem (Power, 1980). In all cases, brook trout have stringent habitat requirements with water temperature being considered a key factor in determining habitat use (McCrimmon and Campbell, 1969; Power, 1980; Barton *et al*., 1985; Meisner, 1990; Flebbe, 1994) because of implications for growth and fitness (Power, 1980;

Hutchings, 2006). In addition to a suitable thermal environment, a year-round supply of clean, well-oxygenated water, adequate in-stream cover and suitable spawning gravels with oxygenated up-wellings of groundwater are all apparent habitat necessities (Power, 1980; Curry *et al*., 1997). Thus streams with cool, quiet pools located between runs of fast water, or rapids, are typical of the stream reaches where brook charr are commonly found in southern Ontario.

Historical evidence indicates that brook charr stocks have changed considerably in the last 100 years, with declines in size and abundance being among the more obvious changes (Power, 1980; Hudy *et al*., 2008; D'Amelio and Wilson, 2008). While studies have implicated the effects of heavy exploitation that select against rapid growth and for early maturity as being primarily responsible (Power, 1980; Nuhfer and Alexander, 1994), early workers noted that populations in small streams became sexually mature at an early age, never attained large size and appeared distinct from forms found in larger rivers and lakes (Power, 1980).

Historical and current land use practices have combined to largely restrict these distinctive small stream populations to isolated headwater streams scattered throughout the southern portion of the former distributional range (Hudy *et al*., 2008). Many of these streams continue to be threatened in southern Ontario, and elsewhere, by stressors such as high water temperature, agriculture, urbanization, exotic fish species, and poor riparian habitat often viewed in isolation by planners, fisheries managers, or the public as not being significant for resident fish species. Cumulatively such changes can result in extensive environmental changes and/or reductions in population size (e.g., Power and Power, 1995). For example, in Southern Ontario agricultural and urban development has removed much of the original Carolinian forest cover that facilitated the existence of, and connectivity between, brook charr populations. As a result, brook charr abundance has been reduced and their habitats are now restricted to a relatively small number of headwater reaches (Blackport *et al*., 1995) where forests and wetlands remain to produce the required groundwater dominated habitats (Barton *et al*., 1985; Bowlby and Roff, 1986; Witzel and MacCrimmon, 1983). Coupled with the noted distinctiveness of form and life-history in small streams, the restriction of brook charr populations to fragmented habitat patches suggests that many southern populations may have significant conservation value.

One of the most important features of the remaining habitats for brook charr is the groundwater that can influence the distribution, reproductive success, biomass and productivity, behaviour and movements of fishes in general (Power *et al*., 1999), and which is critical for the maintenance of healthy brook charr habitats because it facilitates the seasonal continuance of coldwater baseflows that help maintain habitat suitability (Blackport *et al*., 1995). Thus, groundwater extraction with the potential to reduce baseflow correspondingly threatens to reduce overall habitat availability/suitability and may alter stream thermal regimes in ways that will reduce brook charr growth, fecundity and/or survival (e.g., Power, 1980; Hutchings, 2006). Unfortunately, the effects of water extraction on stream fish populations in general, and brook charr populations specifically, has received little attention in the scientific literature (Blackport *et al.*, 1995), with the consequence that it is currently not possible to accurately estimate either the nature or magnitude of water extraction impacts on affected populations.

In 2003, a two-year study was begun to investigate the potential effects of groundwater extraction on brook charr using a three well-site pumping site adjacent to Monora Creek that supplies water to the Town of Orangeville. Overall, the objectives of the study were to determine the effects of pumping from the wells on stream habitat and the resident brook charr population. Specifically, the study aimed to test the hypotheses that: (1) well-related water extraction was associated with habitat degradation at sites proximate to the extraction point; and, (2) degraded habitat was associated with reduced juvenile and adult brook charr use and preference, reduced density and reduced condition.

Methods

The Monora Creek study site is located at 43^{o} 55’ 40” N and 80^{o} 7’ 10” W adjacent to the Town of Orangeville, Ontario (Figure 4.1). The stream is a first order, spring-fed tributary of the Credit River whose flow is supported by an upstream complex of wetlands forming part of a provincially significant wetland known as the Orangeville Wetland Complex. White cedar (*Thuja occidentalis*), balsam fir (*Abies balsamea*) and yellow birch (*Betula alleghaniensis*) dominate over most of the study site. Where present, the ground flora consist of wetland obligate species, including species indicative of groundwater seepage (e.g., kidney-leaved violet, *Viola renifolia*, and curly horsetail, *Equisetum scirpoides*). The creek watershed area is approximately 1.8 km^2

and the creek flows for approximately 3 km before discharging into Island Lake. Downstream of the study site, a storm water retention pond prevents upstream migration of other fish species. Thus brook charr are the only known fish species in the upper reaches of the stream.

Watershed geology is characterized by thick deposits of ice contact drift, primarily sand and gravel, mixed with silty sand till (Cowan, 1976). A set of community water wells (Dudgeon) were drilled in an area of outwash sand and gravel that provides connectivity to creek surface flows. The natural head at two of the three wells (8A, 8B) is artesian (Ontario Ministry of Environment, 2007). The wells draw their water from the lower part of the Amabel aquifer, located approximately 80 meters below surface.

Three sampling sites were selected along Monora Creek in proximity to the well extraction sites (Figure 4.1). A zone of influence, defined as the area over which the hydraulic gradient changed during experimental pumping was defined in 1997 with a shutdown test (Freeze and Cherry, 1979) conducted as part of a provincially mandated study for water extraction permitting. Two sites, A and B, located outside the zone of influence, were selected as reference sites. Sites were approximately 50 m in length and separated by approximately 200 m. Given the size of the zone of influence (≈200 m upstream and 250 m downstream) sample sites within the zone of influence could not be isolated by 200m without the introduction of confounding influences from road runoff occurring immediately downstream of the wells. Thus a single study site was used for the zone of influence.

Stream Habitat and Morphology

Sampling site stream channel habitats (Table 4.1) were classified on the basis of hydraulic and morphological characteristics into channel geomorphic units following Bain and Stevenson (1999). Riffles consisted of areas of shallow, turbulent water with uniform substrate composition and cross-sectional depths <0.2 m. Pools consisted of slower moving water with a visible sorting of substrate and cross-sectional depths >0.2 m. Runs consisted of shallow, slow waters with uniform cross-sectional depths of <0.2 m.

Riparian vegetation percent cover adjacent to study sites was assessed visually, with cover densities clumped into 10% increments. Undercut banks were inventoried. Undercut banks

were defined as stream bank areas with visible overhang of >10 cm where the bank has been cut away by the flowing water. Stream substrate was classified as fines, gravels and cobble using a modified Bovee scale (Bovee, 1982), with fines including all substrate particles ≤4 mm and gravel and cobble, respectively being in the 4-64 mm and >64 mm particle size range.

Stream channel characteristics were measured using a transect methodology (Isaak and Hubert, 2000). Transects were spaced every 5m and wetted width (W) was measured to the nearest centimetre. Water depth (D) was measured at one quarter, one half, and three quarters of the wetted width transect. Habitat area (m^2) was calculated as the product of the length (m) and the mean width (m) of study reach transects. Habitat volume (m^3) was computed as habitat area multiplied by mean water depth (m).

Groundwater Characteristics

Eight pairs of piezometers (Freeze and Cherry, 1979) were installed in 1996 as part of mandated monitoring, two in each of Sites A and B and four in the impact zone. Each piezometer was placed from one to two metres below the stream bed to measure hydraulic gradient, defined as the pressure difference between intergravel and surface water. Piezometers were constructed following a standardized design (Schwartz and Zhang, 2003). Piezometer measurements were taken each month by staff at the Town of Orangeville and at all study sampling dates to determine hydraulic gradient (i) calculated as (Lee and Cherry, 1978):

$$i = (h2 - h1)/d \quad (1)$$

where h1 and h2, respectively, are the downstream and upstream heads measured in metres and d is the distance (m) between the piezometers. Gradient averages were calculated using data obtained over the period April 1, the assumed date of brook charr fry emergence from streambed gravels (e.g., Curry *et al.,* 1991), to a given date of fry capture.

Hydrologic Characteristics

Stream flow measurements made in the late July through to the end of August period 2003 to 2004 were used to characterize baseflows in Monora Creek. In all instances,

measurements used to compute baseflow were taken at the end of periods (>72 hours) over which there had been no recorded precipitation, as such measurements best represent baseflow conditions (Herschey, 1978).

Water velocity (V) was measured ($m^3 s^{-1}$) with a Marsh-McBirney model 2100 Flo-mate (Marsh-McBirney Inc, Fredrick, Maryland, USA) electromagnetic current meter every 0.1 m across the wetted width of each channel survey transect (Riggs, 1985). Velocity measurements were taken at 0.6 of the water depth to obtain the average velocity for each transect-point (Herschey, 1978; Riggs, 1985). Water depth was measured at each of the transect points to the nearest 0.1 cm on a meter stick immediately before the velocity measurements were made. Discharge (Q) was calculated as the total volume of water flowing past the transect per second ($m^3 s^{-1}$), using the equation:

$$Q = WDV \quad (2)$$

Environmental Variables

Daily air temperatures and precipitation data for 2003 to 2004 were obtained from the Environment Canada weather station in Orangeville (ID# 6155790), located approximately 3 km from the stream study site. Water temperature was also measured at Site A and the Impact Site every 30 minutes using Onset HOBO (Onset Computer Corporation, Pocasset, Massachusetts) instream data recorders. The recorders were deployed in the middle of each study section approximately 5-10 cm off the stream substrate to reduce the potential confounding thermal effects of any groundwater upwellings. The recorders were placed at each site in April 2003 and retrieved in November 2004. Data were downloaded from each site on a regular basis, approximately every four weeks.

Fish Capture

At Sites A and B a 50 m section of stream was isolated with blocking nets and a Smith-Root 12B-POW (Smith-Root Inc., Vancouver, Washington, USA) backpack electro-fishing unit was used to capture fish. The depletion method (Zippin, 1958) was used to estimate the abundance of age-0 brook charr in the summer (July) and fall (September-October) of 2003 and

2004 based on three pass capture totals (see below). Electrofishing effort was standardized by thoroughly searching all stream habitats during each pass and with the use of the same electrofisher operator throughout the study. Approximately equal effort, as measured by electrofishing seconds, was used on each pass. The electro-fisher was operated with pulsed direct current at 60 pulses/sec and on a 10% duty cycle. Peak voltage was generally set at 350 V, but was adjusted (range=325-400 V) to maintain fishing effectiveness according to the mean channel width and depth of each study site. Reaches were further left undisturbed for a minimum of 30min between electro-fishing passes to allow for disturbance recovery. The sampling design minimized electro-fishing and temperature-induced changes in fish behaviour that might otherwise violate the assumption of constant catchability employed by closed-population removal estimators (Zippin, 1958).

Captured fish were measured to the nearest 1.0 mm (fork-length, FL) and weighed to the nearest 0.1 g. Due to permit restrictions on lethal sampling and handling, literature-based length-frequency histograms were used to separate fish into age-0 and older age categories (e.g., McCrimmon, 1960; McFadden, 1961; McFadden *et al.,* 1967; Power, 1980; Power and Power, 1995). In the Fall of 2003 and 2004, scales suitable for age determination in short-lived populations of brook charr (Dutil and Power, 1977) were removed from a permitted subsample of captured fish to validate the length-frequency age-group separation. Validation was completed using linear regression to test, respectively, for a zero intercept and unity slope in the relationship between length-frequency assigned and scale determined ages.

Length data from each site were used to model growth patterns using a sine wave von Bertalanffy growth model (Pitcher & MacDonald, 1973) estimated as:

(3) $$L_t = L_{MAX}(1 - e^{-K_1})$$

where,

(4) $$K_1 = C\sin\left(\frac{2\pi(t - s)}{52}\right) + K(t - t_0)$$

Where, L_{MAX} is the asymptotic length towards which fish grow, L_t is the length of the fish at age t, K_1 is the von Bertalanffy growth rate parameter (K) modified by the addition of a sine function expressed in terms of weeks, C and S are constants controlling the magnitude of sine wave oscillations and the starting point for the sine wave, and t_0 is the hypothetical age-at-zero length.

Age-0 and adult length data were also used to estimate mortality (M) using a median length based estimator (Hoenig *et al.*, 1983):

$$Z = 0.693 / (Y_{MEDIAN} - Y_X) \quad (5)$$

Where, Z is the instantaneous rate of mortality ($M=1-e^{-Z}$), $Y_{MEDIAN} = -\ln(1- L_{MEDIAN} / L_\infty)$, L_{MEDIAN} is the median length of fish captured above length L_X, the length at which fish are equally vulnerable to the capture gear (Miranda and Bettoli, 2007), $Y_X = -\ln(1- L_X / L_\infty)$ and L_∞ is the asymptotic length of fish as estimated from a standard single season von Bertalanffy growth analysis (e.g., Wootton, 1998). Significant differences among estimates were established by construction of 95% confidence intervals using the approximate variance for Z as described in Hoenig *et al.*, (1983).

Population Density

The numbers of fish captured on each electro-fishing pass were used to estimate stream population abundance using the three pass maximum likelihood estimator of Junge and Libosvársky (1965).

$$N_t = \frac{6X^2 - 3XY - Y^2 + Y(Y^2 + 6XY - 3X^2)^{0.5}}{18 (X - Y)} \quad (6)$$

Where, N_t is the population at time t, $X = 2y_1 + y_2$, $Y = y_1 + y_2 + y_3$ and y_1, y_2, y_3 are respectively, the number of fish captured in the first, second and third samples. Abundance estimates were divided by sample area to obtain density estimates (e.g., numbers of fish per 100 m^2) to standardize for differences in sample site areas. The constant probability of capture assumption of the Zippin method was tested using a chi-square based statistic as described in Seber (1982).

Statistical Analyses

All statistical analysis were performed using the statistical program STATISTICA (version 8.0). Significance in all statistical procedures was judged at the $\alpha = 0.05$ level. The statistical significance of any differences among stream morphology, hydrology and environmental variables was determined with ANOVA followed by Tukey's HSD post-hoc test, after establishing variance homogeneity (e.g., Levene's test). Differences in the age-0 and adult biological variables: fork-length, weight and density were similarly tested. Inter-annual comparisons of mean daily air temperature, precipitation and water temperature data were also completed using ANOVA. Fisher's exact test was used to test for significant differences in the proportion of juvenile and adult captures in the reference and impact sites. Linear regression was used to estimate weight-length relationships (Wootton, 1998) for the data obtained each year at each study site and regressions were assessed for statistical adequacy using standardized error term tests (Zar, 1999). The prevalence of isometric versus allometric growth was assessed by testing for significant differences between the estimated weight-length regression slope coefficients and the theoretical value of three that indicates isometric growth (Wootton, 1998). Differences in the estimated weight-length relationships by year at each study site, and among study sites, were tested using ANCOVA (Zar, 1999).

Standard and sine wave von Bertalanffy models were estimated using non-linear regression routines in STATISTICA and residuals examined for statistical adequacy as described in Bates & Watts (1988). Significant differences between estimated von Bertalanffy models were examined using analysis of residual sum of squares (Chen *et al.*, 1992; Haddon, 2001).

To test for habitat selection preferences among the study sites, sample site areal estimates were converted to proportions (P_i) and compared to proportionate use values (o_i) as determined from fish capture data with standardized selection indices (Manly *et al.*, 1993). Standardized ratios (β_i) were computed by normalizing site-specific selection ratios ($W_i=o_iP_i^{-1}$) computed from the proportional availability and use data for each site. Standardized selection ratios may vary from 0 to 1 and will sum to 1 for all sites considered (Manly *et al.*, 1993). Standardized ratios of n^{-1} indicate no preference. Values $>$ and $<$ n^{-1} indicate, respectively, relative preference and avoidance of the study site (Manly *et al.*, 1993). Selection ratios were tested for statistical significance using the χ^2 based statistic described in Manly *et al*. (1993) under the null hypothesis

that site habitat occupancy occurred at random. Significant differences among computed selection indices were established using Bonferroni-adjusted confidence limits and χ^2 tests of significant differences between standardized selection ratios as outlined in Manly *et al*. (1993).

Results

Channel site morphology, hydraulic gradient, baseflow and habitat features varied among the study sites (Table 4.1), with ANOVA indicating significant differences in wetted widths, water depths and hydraulic gradients (Tukey's HSD $P<0.05$). Channel wetted width increased monotonically in the downstream direction, but was not matched by increases in mean channel depth, baseflow or hydraulic gradient. Hydraulic gradient was significantly correlated with water depth ($r^2=0.78$, $P<0.001$) and surface area ($r^2=0.50$, $P=0.002$) and differed among study sites (ANOVA, $P<0.001$), with negative gradients predominating in the Impact Site and positive gradients dominating in Sites A and B.

Habitat surveys completed coincident with fish sampling revealed differences among the sites, with the most notable difference being a reduction in the % under-cut banks, increase in the % of riffle habitat and reduction in the % of run and pool habitat (Table 1) in the Impact Site. Although there were no appreciable differences in substrate composition, the winter shallow riffle sections in the Impact Site with low hydraulic gradients tended to be dominated by anchor ice formations not observed in Sites A or B.

Mean daily precipitation and air temperatures did not differ significantly, during the study period 2003 to 2004 (precipitation: $F_{1,334}=1.33$, $P=0.25$; air temperature: $F_{1,334}=0.54$, $P=0.46$). There were also no significant differences in the mean daily water temperatures measured by the instream temperature loggers at the reference and impact sites ($F=_{1,670}$, $P=0.08$).

A total of 329 brook charr were captured during sampling in 2003–2004. Percent relative frequency distributions (Figure 4.2) for all fish captured show significant differences in the proportion of age-0 fish (≤100 mm) found in Sites A and B versus the Impact Site (Fisher's exact test, $P= 0.024$). When all fish were combined, there was a significant difference between mean fork-lengths by site (ANOVA: $F_{2,326}=28.68$, $P<0.001$), with mean fork-length at Site A (84.6 mm)

being significantly lower than at Site B (118.1 mm) or the Impact Site (116.3 mm) (Tukey's HSD $P<0.001$).

Length-frequency assigned and scale determined ages compared favourably ($r^2=0.92$, $P<0.001$), with the regression of length-frequency assigned ages on scale determined ages yielding a zero intercept ($P=0.14$) and a slope equal to 1 ($P=0.25$). Mean fork-length at age, standard deviations and sample size are reported in Table 4.2. Among studied sites, fish-at-age generally did not differ in length, with the exception being at age-2 where the mean fork-length of fish in Site B and the Impact Site exceeded that of fish in Site A ($F_{2,74}=3.99$, $P=0.022$). Furthermore, by age, fish do not differ significantly in size between years at a given site ($P>0.05$). The result is consistent with the analysis of residual sum of squares, as performed for the growth model analysis (see below).

In 2003, estimated site-specific weight-length relationships evidenced isometric growth, with statistical tests showing no significant differences between the estimated weight-length slope coefficient and the theoretical value of three indicative of isometric growth (minimum test $P\geq0.252$. In 2004, allometric growth predominated at Site A and the Impact Site, with weight increasing more in proportion to length in Site A (weight-length slope = 3.11, isometric test $P= 0.015$) and in less than proportion to length in the Impact Site (weight-length slope = 2.69, isometric test $P< 0.001$). Among years, ANCOVA indicated no significance differences in the weight-length relationship for Site A ($df=113$, $t=-1.402$, $P=0.164$) or Site B ($df=92$, $t=0.760$, $P=0.449$), but there was a significant difference for the Impact Site ($df=110$, $t=3.320$, $P=0.01$). When data for all years were grouped by site, ANCOVA also indicated no significant difference for 2003 in the slopes of the weight-length regression models estimated for each ($F_{2,158}=1.80$, $P=0.169$), but heterogeneity of the model intercepts ($F_{2,160}=5.15$, $P=0.599$). In 2004 there was a significant difference in the slopes of the weight-length regression models estimated for each site ($F_{2,157}=12.15$, $P<0.001$), but no heterogeneity of the model intercepts ($F_{2,159}=1.96$, $P=0.144$). Mean condition as represented by the estimated weight-length regression coefficient was positively correlated with hydraulic gradient ($r^2=0.63$, $P=0.05$).

Sufficient length-at-age data were collected in 2003 and 2004 to estimate sine wave von Bertalanffy growth models for all sites. Analysis of residual sum of squares indicated there were no significant differences among site-specific models ($F_{15,309}$=0.83, P=0.648) such that the common model could be adequately used to describe length-at-age in all sites (Figure 4.3) as follows:

(7) $$L_t = 422.04\left(1 - e^{-K_1}\right)$$

where

(8) $$K_1 = 0.014 \sin\left(\frac{2\pi(1 - 13.679)}{52}\right) + 3.17x10^{-3}(t + 20.54)$$

Length-based estimates of mortality showed no pattern, with site-specific mortality 95% confidence intervals overlapping for all sites and all ages. Age-O mortalities ranged from a low in 2004 of 0.838 at Site A to a high in 2003 of 0.916 at the Impact Site, with a mean ± standard error of 0.877±0.014. Adult mortalities were consistently lower, ranging from a low of 0.682 at Site B in 2004 to a high of 0.778 at Site A in 2003, with a mean ± standard error of 0.724±0.14.

Testing of the equal probability of capture assumption used to estimate sample site densities indicated the assumption was not violated for any of the 2003 or 2004 site-specific estimates (all test P>0.05). Summer and fall Age-0 densities at Site A were significantly higher than at either Site B or the Impact Site (Figure 4.4). Age-0 densities at Site B and the Impact Site did not differ during the summer, but there was a significant difference in the fall when densities at Site B exceeded those at the Impact Site. With the exception of Site B in the fall, there were generally no significant differences among site adult densities. No significant correlations between measured habitat variables and density were found.

Standardized selection indices computed from proportional habitat availability and use data indicated significant differences among the sites with respect to the preferences of age-0 fish (Table 3), with age-0 fish showing a significant preference for Site A in all seasons (P<0.05). In addition, Site B was preferentially selected for occupancy in the fall of 2004 (P<0.05). Adult fish showed no significant differences in preferences for any of the study sites in the summer

(P>0.05), but by fall there was a distinctive and significant preference for Site B in both 2003 and 2004 (P<0.05). In contrast preferences for Site A and the Impact Site were not significantly different (P>0.05). Within years, 95% confidence for site selection preferences indicated no significant differences between summer and fall for both age-0 and adult fish.

Discussion

Study results indicate significant, although subtle, changes in available habitat as a result of pumping activity, with habitats in the Impact Site showing significant decreases in hydraulic gradient, stream depth and the availability of under-cut banks and run and pool habitat. Combined, the changes in habitat influenced use, with age-0 densities declining in the Impact Site relative to the non-impacted Sites A and B. When adjusting for relative availability, standardized preference indices indicated a significant preference for non-impacted Site A among age-0 brook charr. Adults evidenced slight changes in distribution among the available sites, with reduced densities in the impacted site and preference for non-impacted habitat in Site B in the fall. Changes in condition (weight-length relationship) and mortality were generally not in evidence during the course of this study. Thus overall, the impacts on brook charr populations subjected to moderating pumping appear to relate more to patterns of within stream distribution and habitat use than to any direct measures of fitness such as survival or growth.

There are numerous studies (Armstrong *et al.,* 1998; Bain *et al.,* 1988; Covington, 2003; Curtis, 1959; Cushman, 1985; Curry *et al.,* 1994; Glova and Duncan, 1985; Kraft, 1972; Nuhfer, 1998; Strevens, 1999; Wesche, 1974; Wipperman, 1967) dealing with the effects of water taking on physical habitat, and several studies documenting the effects of flow reductions on metrics of fish population health (i.e., growth rates, stress responses) for populations exposed hydroelectric dam operations (Bain *et al.*, 1988; Arnekleiv *et al.*, 2004; Flodmark *et al.,* 2002). However, there has been relatively little research on the effects of flow reductions associated with limiting groundwater additions, as might occur when groundwater is pumped (Latta, 1965; Benson, 1953; Strevens, 1999). The studies that do exist point to correlations between juvenile densities and groundwater flow. Latta (1965) noted that during years of high groundwater levels, there were larger numbers of age-0 brook charr in study sections of the Pigeon River, Michigan, but that adults were not similarly affected. Kraft (1972) similarly noted the redistributive effects of

moderate flow reductions in a small Montana stream, and observed no overall changes in trout abundance because of the tendency of brook charr to redistribute themselves to areas with higher flow.

The most obvious and direct impact of pumping was on fish habitat via a reduction in stream channel hydraulic gradient, a measure of the groundwater baseflow made available to the surface channel, and the avoidance of low hydraulic gradient habitats by juvenile brook charr. Studies of brown trout (*Salmo trutta*) in Dorset, U.K., chalk streams have noted similar impacts with a spatial correlation between a zone of reduced river flow and an area of low juvenile trout abundance confined to the critical May to October juvenile growth period (Strevens, 1999). Associated simulation-based (PHABSIM) studies have further suggested reduced impacts on weighted usable areas available for adults as compared to juveniles, particularly at low discharges (Strevens, 1999). Differences in the relative impacts of flow reductions on age-specific weighted useable areas would predict the statistically insignificant variations in observed adult densities recorded in this study. Drought studies have also shown juvenile salmonids to be more sensitive and disproportionately influenced by summer drought low flow conditions as a result of reductions in the availability of suitable habitat (Elliott *et al.*, 1997).

Age-0 brook charr generally occupy shallower and slower reaches than older fish (Bain *et al.,* 1988). On the basis of similar depth and velocity characteristics alone there is no *a priori* reason to suggest occupancy of the Impact Site would not approximate what was observed at the other study sites. Studies of drought-induced movement of Atlantic salmon (*Salmo salar*) juveniles suggest fish may emigrate simply in response to the direct adverse local stimuli represented by low flows (Armstrong *et al.*, 1998) or, in this instance, the loss of groundwater inputs know to favour juvenile survival, particularly in winter (Power *et al.,* 1999). For juvenile salmonids dispersal is known to be influenced by inter- and intra-specific competitive factors and habitat quality such that the propensity to move in response to changes in flow may be related to size (Scruton *et al.,* 2003). For example, territorial affinity may preclude dominant, larger fish from easily abandoning acquired territories (Mäki-Petäys *et al.,* 1999). Conversely, Hutchings (1986) has noted that if competition leads to migration and the dominance hierarchy is size dependent, smaller individuals would be those expected to move. In the context of the Monora Creek study, the above suggest that mechanisms for the reduction in habitat occupancy densities

in low hydraulic gradient habitats are likely to include territorial behaviour and increases in agonistic competition necessitated by the reduction of habitat area (Bain *et al*., 1988; Strevens, 1999).

Shirvell (1994) reported that movement in response to changing stream flow appeared to be voluntary, and was often in an upstream direction, as was found here. Most subsequent studies have reported similar responses, whether in the context of the use of experimental peaking flows (Scruton *et al.,* 2003) or experimental trials (Armstrong *et al.,* 1998; Huntingford *et al.,* 1999; Davey *et al.,* 2003). Generalizations among species, however, may be difficult if impact disturbance is contingent upon species-specific refuge-use strategies likely to result from interplays between refuge availability, disturbance characteristics and fish behaviours and morphologies (Davey *et al.,* 2003). Nevertheless, losses of functional habitat heterogeneity have been associated with significant changes in selection preferences reflected within stream distributional shifts (e.g., Bain *et al*., 1988). For any specific location in a stream, changes in stream flow will translate directly into changes in water depth and velocity that influence stream substrate and instream cover, with associated changes in habitat heterogeneity that have implications for fish abundance that are dependent on age (Bain *et al.,* 1988). Thus, while loss of groundwater baseflow appears to have immediate impacts on age-0 brook charr, the impacts on older fish were not apparent until autumn when seasonal flows naturally begin to approach a minimum (Herschey, 1978) and potential loss of under-cut bank habitat becomes more critical.

Although findings here indicate subtle, but important, impacts on brook charr populations associated with moderate groundwater extraction, uncertainties remain about the point at which extraction will result in severe reductions in abundance. Results here indicate groundwater extraction can affect key stream hydraulic characteristics, affect stream morphology and induce differential use of stream habitat. Analogue studies of drought impacts further suggest that severe reductions in flow significantly reduce the abundance of all age-classes of brook charr (Hakala and Hartman, 2004). While uncertainties remain about the precise nature of the relationships between groundwater flow and the sustainability of headwater brook charr populations, it is now possible to identify some of the mechanisms that result in populations impacts. Monitoring of redistributional effects will permit early evaluation of the potential impacts of reducing baseflow

contributions to headwater streams and such considerations should be incorporated into decisions governing the selection of alternative groundwater pumping locations.

Table 4.1. Study site mean stream morphology, baseflow, hydraulic gradient and habitat features, based on April 1 to September 15, 2003-2004 measurements. Substrate bed materials were classified to size following a modified Bovee (1982) scale. Among site differences in means are categorized with alphabetic superscripts, with means that are not significantly different sharing a common superscript. Among year differences for each study site are categorized with numeric superscripts, with superscripts indicating patterns of yearly differences as follows: 1 – 2003 > 2004 and 2 - 2004 > 2003. C of V % defines the among site coefficient of variation in percent terms for each habitat feature.

							Substrates			Channel Geomorphic Units		
Sites	Wetted Width (m)	Water Depth (m)	Baseflow (l/s)	Hydraulic Gradient (i)	Riparian Vegetation % cover	% Under-cut banks	% Fines	% Gravels	% Cobble	% Riffle	% Run	% Pool
A	1.11 ^{A}	0.153 ^{A}	9.9	0.033 ^{A}	75	40	40	50	10	10	40	50
B	1.54 ^{B}	0.202 ^{B}	13.7	0.279 ^{B}	75	50	70	25	5	5	35	60
Impact	1.91 ^{C}	0.130 $^{A, 1}$	13.9	-0.268 $^{C, 2}$	75	0	30	50	20	60	20	20
C of V %	26.3	22.7	18.0	-	0.0	88.0	45.0	34.6	88.0	122.0	33.0	48.0

Table 4.2. Mean fork-length (mm) at age ± standard deviation and sample size (n) of brook charr captured at study sites, 2003-2004. Within age categories, superscripts denote like means as determined using a one-way ANOVA. ANOVA P defines the p-value associated with each test.

Age	Site A	n	Site B	n	Impact	n	AVOVA P
0	50.0 ± 14.6 ^{A}	48	58.6 ± 11.5 ^{A}	17	52.5 ± 16.3 ^{A}	13	0.107
1	101.4 ± 22.5 ^{A}	60	108.2 ± 15.4 ^{A}	46	105.7 ± 16.1 ^{A}	63	0.156
2	143.6 ± 15.5 ^{A}	11	158.4 ± 14.7 ^{B}	30	153.6 ± 14.8 $^{A B}$	36	0.023
3	NF		202.7 ± 4.9 ^{A}	3	194.0 ± 5.7 ^{A}	2	0.165
Total		119		96		114	

Table 4.3. Standardized selection preference ratios for age-0 and adult brook charr by study site, season and year. Preference ratios not significantly different from one another for a common season and year are denoted with a common superscript.

Time Period	Site A		Site B		Impact	
	Age-0	Adult	Age-0	Adult	Age-0	Adult
Summer 2003	0.700 [A]	0.401 [A]	0.076 [B]	0.330 [A]	0.224 [B]	0.269 [A]
Fall 2003	0.541 [A]	0.269 [A]	0.338 [A]	0.439 [B]	0.121 [B]	0.292 [A]
Summer 2004	0.887 [A]	0.400 [A]	0.113 [B]	0.326 [A]	0.000 [B]	0.274 [A]
Fall 2004	0.670 [A]	0.319 [A]	0.300 [B]	0.424 [B]	0.030 [B]	0.257 [A]

Figures

Figure 4.1. Location of study sites in southern Ontario.

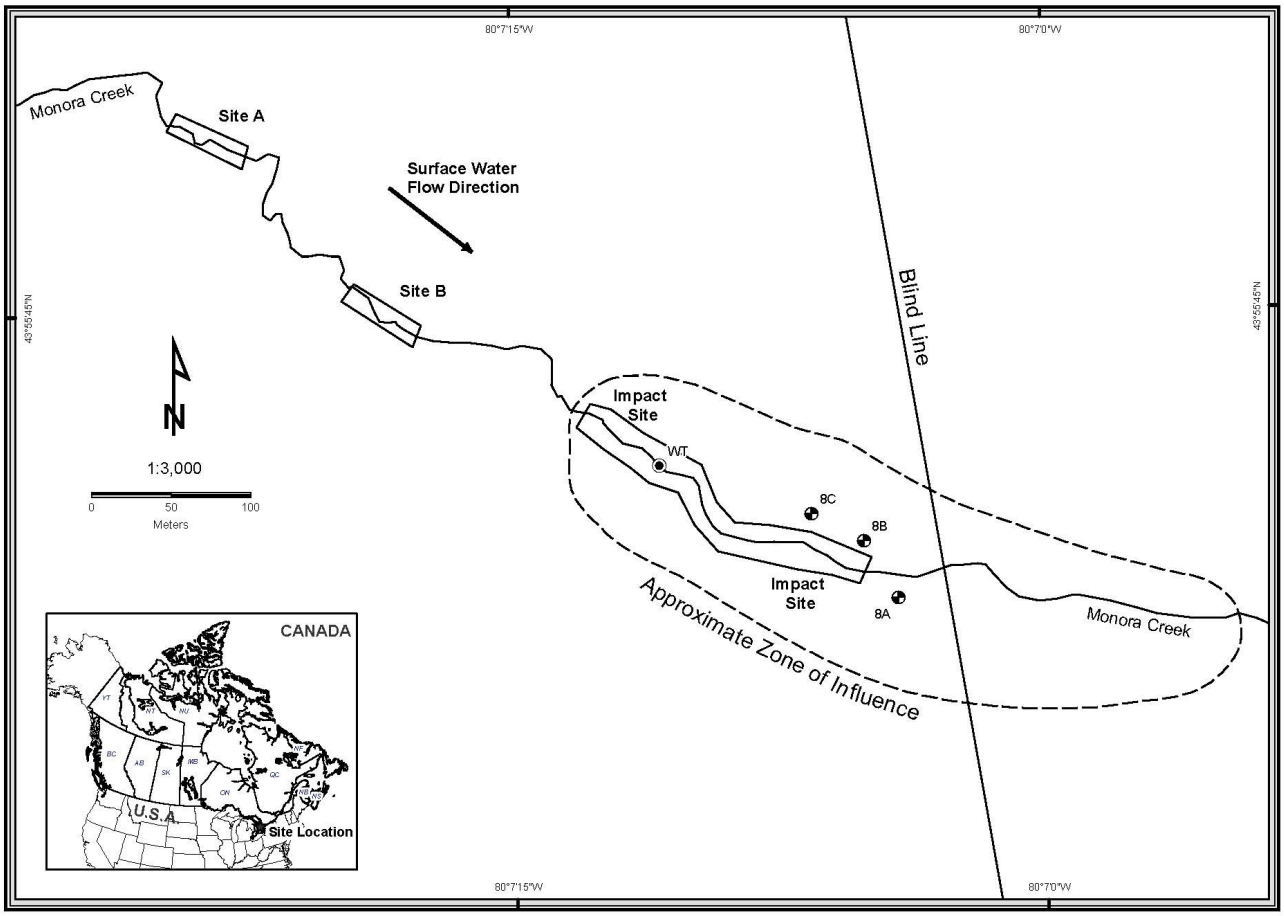


Figure 4.2. Brook charr length - frequency distributions by capture site in Monora Creek, 2003-04.

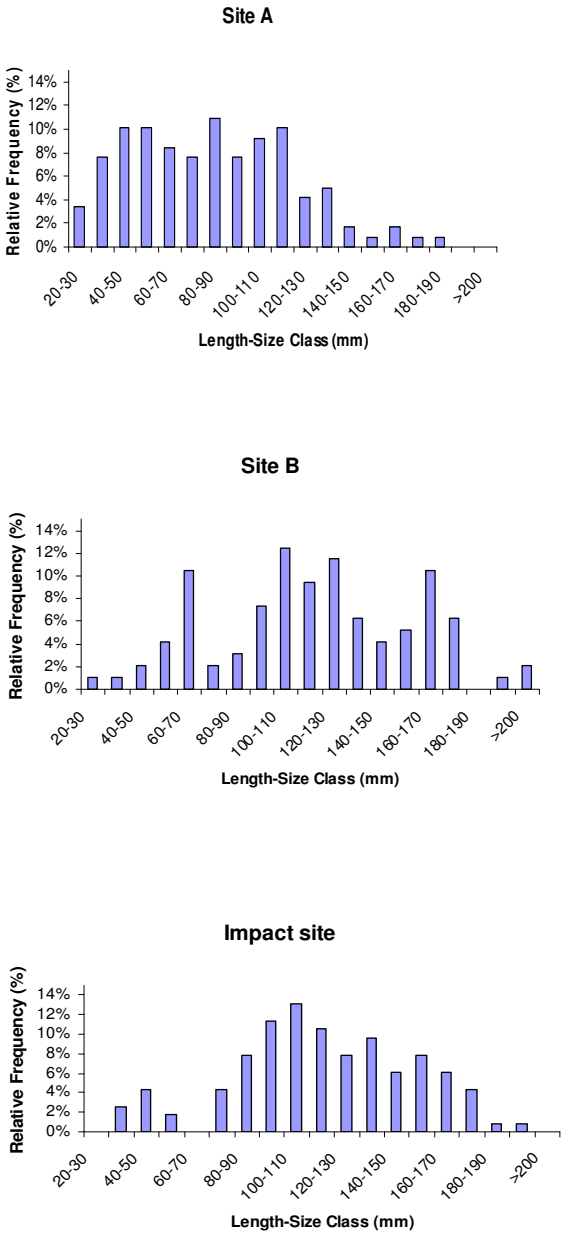


Figure 4.3. Site specific fork-length by age (weeks) of brook charr from in Monora Creek and estimated sine wave von Bertalanffy growth model (solid line). Data for fish captured in non-impact Sites (A and B) and the Impact Site, respectively, are plotted as dark and white circles.

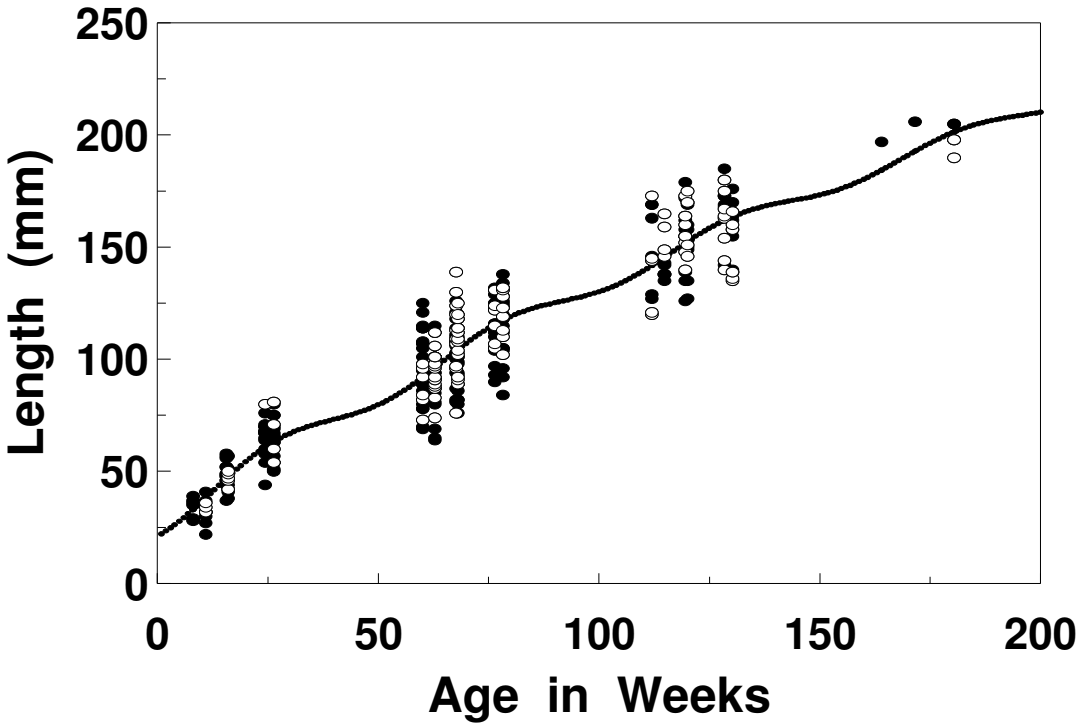
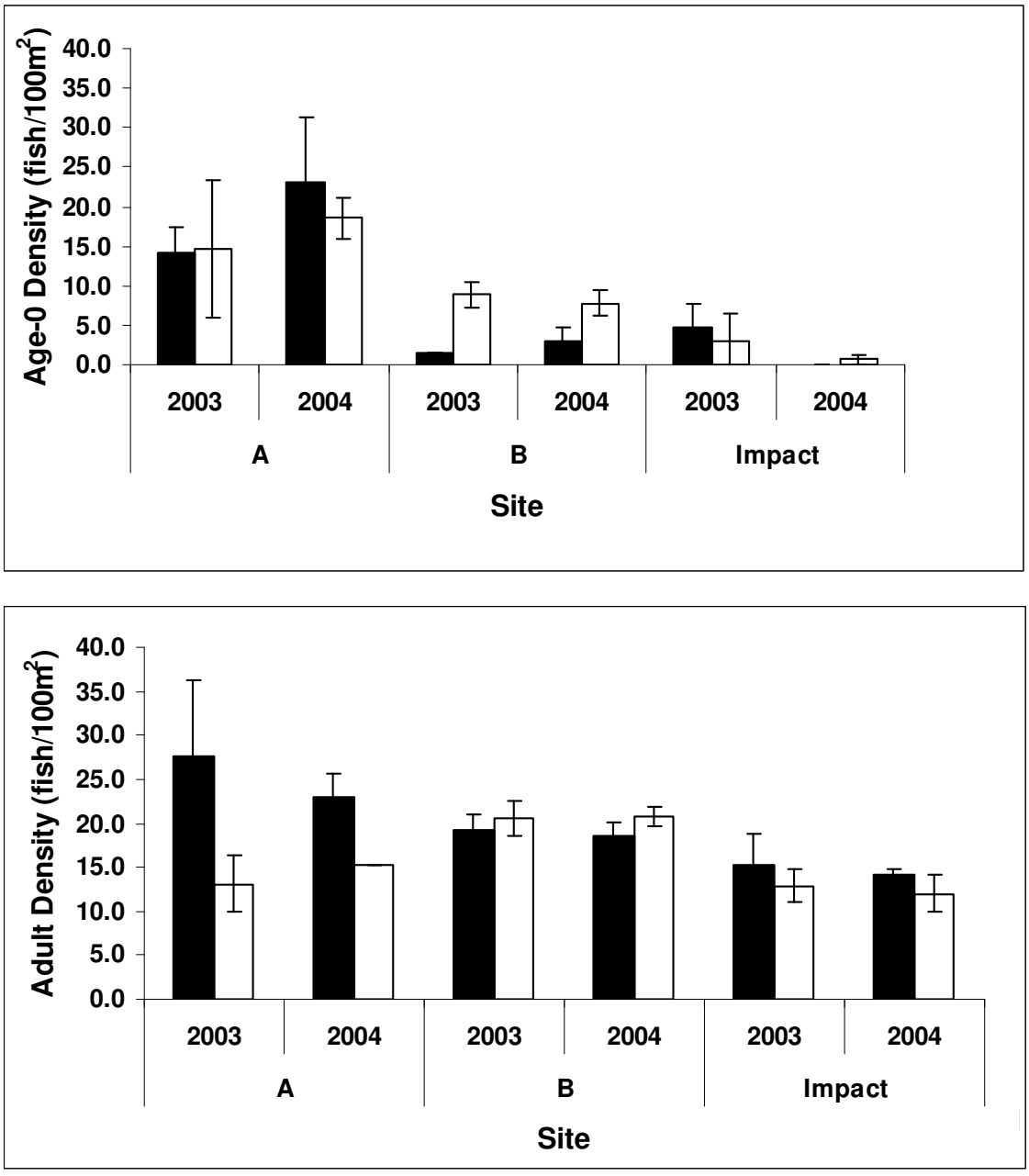


Figure 4.4. Age-0 and adult density estimates with 95% confidence limits for brook charr captured in the summer and fall at Monora Creek study sites, 2003-2004. Data for summer and fall are plotted, respectively, as solid black and white bars.



CHAPTER 5. GENERAL CONCLUSIONS

This thesis investigated the factors influencing the fork-length and density of age-0 brook charr in three headwater streams (Chapter 3), and the potential effects of groundwater extraction on the stream habitat and the resident juvenile and adult brook charr of Monora Creek (Chapter 4).

In Chapter 3, differences in temperature, stream site and flow were found to significantly influence differences in age-0 brook charr length and density, with the effect of temperature being the most significant factor. Although, flow had a pervasive influence on age-0 length and density, the affects were dominated by other physical and biological features of the study sites. While water temperature is known to critically influence brook charr growth, temperature alone will not permit increased growth unless matched by suitable increases in ration. The positive correlation between ATU and capture size, therefore, suggests sufficient matching of temperature-driven increases in metabolic demand and ration, even in streams marginally affected by water extraction.

Possible improvements to data collection might include use of an additional flow station at each study site to improve the estimation of any relationships between flow and age-0 length and density. Specifically, a flow station with a continuous data logger at the upstream (i.e., greatest impact) and downstream (i.e., least impact) end of the study reach would have better captured the variation in flow throughout the study reach. Similarly, deployment of more mini-piezometers would have helped improve characterization of variation in the hydraulic gradient across the study site. In addition, riparian vegetation and instream macrophytes at all three sites made capture of fish difficult with a two person crew. Having two netters and one person electrofishing might have improved our capture rate and decreased the confidence limits around density estimates.

In Chapter 4, groundwater extraction was found to have a significant, although subtle, influence on available brook charr habitat, with habitats in areas affected by groundwater pumping showing significant decreases in hydraulic gradient, stream depth and availability of run and pool habitat. Combined, the changes in habitat influenced use, with age-0 densities declining

in the groundwater pumping affected areas of the stream relative to the non-impacted areas. When adjusting for relative availability, standardized preference indices indicated a significant preference among age-0 brook charr for areas of the stream not affected by groundwater pumping. The impacts on brook charr populations subjected to moderating pumping, therefore, appear to relate more to distribution and patterns of habitat use within the stream than to direct measures of fitness (e.g., survival).

Combined, these studies completed as part of this thesis suggest development actions moderating either the thermal or flow regime of streams will have negative effects on brook charr population status. Water extraction risks altering habitat in unfavourable ways by interrupting important linkages between flow and temperature that favour growth and density, but that prevent critical physiological thresholds for temperature from being surpassed. Furthermore, the impacts on brook charr populations subjected to moderating pumping appear to relate more to patterns of within stream distribution and habitat use than to any direct measures of fitness such as survival or growth. The latter result suggests monitoring abundance and/or condition alone will not suffice to provide early warning of population stress responses to water extraction activities. However, when compared to the potential impacts from other development activities found in southern Ontario watersheds (i.e., subdivision, agricultural, roads), water extraction activities tend to occur in protected forested areas, which have obvious mediating benefits in terms of maintaining a healthy environment for brook charr. Accordingly, low levels of water extraction may be of overall less concern for the sustainability of headwater brook charr populations than other anthropogenic perturbations on headwater streams.

Recommendations and implications for future study

A difficulty encountered during this research was finding enough suitable study sites, where a self sustaining headwater population of brook charr existed and where the only potential impact was water extraction. Many populations occur in small, multiply stressed reaches of stream. When suitable sites were located, it was often challenging to convince the owners to become a research project partner. Owners were generally concerned that if the research showed water extraction adversely affected fish habitat, they would not obtain, or loose, a permit-to-take

water. The combination of site suitability and owner concerns combined to reduce the feasible set of study sites to the small number included in this study.

The research that was conducted could have been improved by: [1] creating more stress on each study system by increasing the pumping rate to further impact the habitat by reducing flow within the studied stream reaches, [2] use of additional hydrologic and hydrogeological stations, [3] increasing site replication, and [4] increasing the number of fish captured and measured on each sampling date. Early collaboration and the active involvement with government regulatory agencies could have addressed those problems. While approaches to agencies were made, most showed no interest in becoming involved in the research producing the data needed to drive regulatory decision-making. As an example, Nassaw Springs provided an ideal location for increasing the stress on the resident fish as the amount of water being removed from the stream could be easily adjusted and associated habitat responses to the change easily measured. However, in discussions with Fisheries and Oceans Canada (DFO) a proposal for extracting more groundwater that was hydraulically connected to the stream was deemed to represent a potential to cause a harmful alteration, disruption or destruction of fish habitat (HADD) requiring a *Fisheries Act* authorization. Interestingly, while experimental designs discussed offered the potential for gathering factual information upon which scientifically-defensible decisions could be based, permitting practices in place at the time were not based on any scientifically defensible information or method. The missed opportunity was a clear failure of regulatory responsibility and did nothing to ensure continued sustainability of endangered and threaten headwater brook charr populations.

Since groundwater extraction can reduce gradients in a stream (measure of baseflow) and impact fish habitat, more hydrology and mini-piezometer stations are recommended for use in routine monitoring for permitting purposes or in any studies attempting to link water extraction activities to biological responses in monitored populations of brook charr. Hydrologic stations should be established upstream and downstream of the zone of influence and at each fish sample site, if possible. Continuous water level meters should also be installed in the same location to collect data at frequent intervals (e.g., every 15 minutes). The loggers should also be surveyed and a bench mark established to ensure they can be replaced, if necessary, without loss of data. Mini-piezometers should be installed next to established hydrology stations whenever possible.

Lack of replicate sites on the small streams available for study and the permit restrictions that necessitated low sample size prevented use of a number of statistical methods that might have improved mortality estimates (e.g., catch curves, change in ratio methods). Because of the concern for headwater brook charr populations in southern Ontario and the likely increasing number of stressors that will be placed on remaining populations, it is recommended that future studies attempt to partner with DFO well in advance of developing the study design. Inclusion of a certified regulatory agency should permit increases in the number of sample replicates within stream and yield greater amounts of data upon. Greater precision in impact estimates will increase confidence in the nature and magnitude of the estimated effects of water extraction on headwater brook charr populations, thereby serving to increase the scientific defensibility of water extraction permit decisions. Then, and only, then can a suitable balance be struck between the need to extract water for human use and the need to conserve and sustain the marginal populations of brrok charr that still inhabit the isolated headwater stream reaches found throughout southern Ontario.

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